2017

Effectiveness of best management practices to increase infiltration in urban and rural environments

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Effectiveness of best management practices to increase infiltration in urban and rural environments

by

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A thesis submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Major: Environmental Science

Program of Study Committee:
Janette Thompson, Major Professor
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Peter Wolter

Iowa State University

Ames, Iowa

2017

DEDICATION

I would like to thank my committee chair, Dr. Janette Thompson, and my committee members, Dr. Sally Logsdon, and Dr. Peter Wolter, for their patience, guidance, and support throughout the course of this research.

In addition, I would like to thank my undergraduate professor, Dr. Reuben Goforth, for giving me the opportunity to realize my appreciation of fresh and saltwater ecosystems. I would like to thank my parents and sister for their support of my academic advancement, my friends for their constant words of encouragement, and my office mates for their comic relief. Lastly, I want to offer my appreciation to the Department of Natural Resource Ecology and Management at Iowa State University for the support of my academic and professional advancement.

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ABSTRACT

Source-control best management practices (BMPs) have been designed and promoted as flexible alternatives for runoff mitigation in both urban and agriculturally developed landscapes and are likely to become more important given climatic predictions of more frequent and intense rain events. Strategies that incorporate vegetative elements and natural soil water infiltration to reduce runoff delivered to conventional sewer or tile drainage systems and increase groundwater recharge are compatible with other characteristics of urban and agricultural landscapes. However, the rate of adoption of BMPs has been slow as a result of uncertainties about maintenance, effectiveness when incorporated during retrofitting, and long-term benefits that have been under studied. In the first part of this study I examined the efficacy of three common stormwater BMPs in a variety of urban (residential, recreational and commercial) environments. Specifically, I examined bioretention cells, native landscaping (restored prairie), and vegetated riparian buffer practices. In the second component of this study, I examined the similarities and differences in performance for a single BMP, vegetated riparian buffers, when used in both urban and rural landscapes. For both studies, I examined performance capacity based on the spatial extent of each BMP (receiving area) to its subwatershed (contributing area). I also conducted rainfall simulation to measure infiltration, absorption capacity, runoff characteristics and collected soil samples to characterize pollutant accumulation. Among the urban BMPs in the first study, bioretention cells and wooded zones of the buffers had the lowest soil bulk densities, highest infiltration rates, and smaller runoff volumes than did their contributing areas. In the second study, I observed that urban buffers, although generally smaller, had larger practice to contributing area

ratios, indicating that spatial constraints may not diminish buffer effectiveness in these landscapes. Rural and urban buffers demonstrated analogous performance for buffer areas compared to their respective contributing areas. In both landscape settings the buffer areas had the highest infiltration rates and the wooded buffer zones demonstrated significantly greater time-to-runoff compared to their contributing areas. In both studies, I determined that the effectiveness of BMPs observed could be enhanced if their surface area was enlarged, or if they were implemented as clustered practices. Further, my findings suggest that while implementation of these practices is likely to reduce runoff volumes and improve water quality, their performance could be improved using site-specific practice designs rather than following more generic technical recommendations.

CHAPTER 1: GENERAL INTRODUCTION

Precipitation that reaches the earth's surface can either infiltrate into the ground, be stored as soil moisture (and potentially transition back into the atmosphere as evapotranspiration), or enter surface hydrological systems. Water may be delivered to surface hydrological systems through runoff if the soil profile has reached capacity for absorption of water or if the surface is impermeable. Climate change is contributing to alterations of hydrologic systems through increased temperatures and more irregular and intense rain events that are difficult to manage (IPCC, 2013). Runoff can accumulate on surfaces as receiving area capacities (e.g. streams, lakes, sewers) are overwhelmed by increased impervious surface and more frequent flooding events (Moore et al., 2016). Although there is strong certainty that human activity is a dominant cause of observed global warming, in the context of this research, climate change will be considered a "natural" contributor to changes in hydrology. Climate adaptation strategies are influencing the design and retrofit of new and existing infrastructure in many urban areas (Gaffin et al. 2012; Sussams et al. 2015). Further, recent interest in the potential benefits of sustainable practices that mitigate runoff and flooding have led to a number of studies to document both the effects of climate change and the role of these practices in different landscape contexts (Gaffin et al., 2012; Gill et al., 2007; Waters et al., 2003).

The velocity at which precipitation enters the soil profile is known as the soil infiltration rate. Typically expressed in millimeters per second, the infiltration rate is an indicator of water movement into and through the pore spaces of soil (Gupta et al., 1991). Soil water infiltration and storage are necessary for subsequent plant root water uptake and plant growth. Soil texture and structure are physical properties that are important in determining total pore space and pore size distribution within the soil matrix, specifically the air-filled macropores and the water-filled micropores (Craul, 1994). Soil compaction can compress macro- and micropores in soil, leading to a reduction in both infiltration and soil drainage capacity (Craul, 1994). Water delivered at a precipitation rate that exceeds the infiltration capacity of compacted soil will pond or flow over sloping land as stormwater runoff (Gregory et al., 2006). Stormwater runoff has been recognized as a primary source of pollution and degradation of surface and below-ground water resources as it conveys sediment and soluble organic and inorganic pollutants to receiving areas (USEPA, 2005).

Human land use usually diminishes the natural capacity of the land surface to absorb water and may also modify the characteristics of hydrological systems. Specifically, urban and agricultural site developments are major contributors to changes in hydrological regimes. Site attributes such as slope, hydraulic conductivity, and surface cover are susceptible to alteration through tillage, traffic of heavy equipment, soil excavation, mixing and regrading, and other physical alterations (Olson et al., 2013; Philips and Kitch, 2011). These site modifications typically also cause soil compaction, decreased permeability, and decreased water infiltration leading to higher peak flow rates and greater need for pollution control in both urban and agricultural environments (Gregory et al., 2006).

Urban land uses are considered the second largest contributor to surface water impairment at a global scale (Paul and Meyer, 2001). Rapid urbanization began in the mid-19th century as economies shifted from agriculture to manufacturing (Burns et al., 2015). These centralized hubs were created in effort to support communities, government, and industry with the convenience of infrastructure, goods, and services delivered from fertile farmland by waterways (Boustan et al., 2013; Foley et al., 2005). Accordingly, the landscape quickly changed from natural land cover types to impervious surfaces. During more recent large-scale residential development, typical site preparation practices strip away topsoil and regrade the land surface with homogeneous engineered fill material (Randrup, 1997).

Agricultural land uses can also have adverse impacts on landscape hydrologic regimes. Since the middle 1800s, private landowners have farmed the Upper Midwest region, over time adopting practices such as extensive tile drainage and frequent application of inorganic fertilizers to increase productivity (Fausey et al., 1995). Today, fertilizers, pesticides, and livestock manure are recognized as major impairments to surface waters (Fuhrer, 1999). Subsurface tile drains, grassed waterways, and compacted cropland soil surfaces expedite polluted water runoff that carries soluble nutrients to surface waters. Enrichment of nitrogen and phosphorus in freshwater systems in particular depletes oxygen, creating hypoxic zones harmful to aquatic life (Vitousek et al., 1997). Sediment is also transported across land via water runoff and can prevent infiltration by clogging soil pores, and promote erosion of streambanks (Angelo, 2013).

In urban environments storm sewer systems typically convey runoff directly to nearby surface waters in order to protect human health and property, but this also exacerbates pollutant inputs and hydrologic disturbance to receiving systems (Roy et al., 2008). Human activities in urban areas also impact the quality of stormwater through pollutants associated with fertilizer application, motor vehicles and industry, and waste management, all of which create macro-elements that can be easily conveyed to surface waters during precipitation events (USEPA, 2005). Systematic stormwater management strategies have been proposed for new development, such as low impact development (LID) that aims to manage stormwater near the source. Vegetated best management practices (BMPs) such as filter strips, swales, wetlands, and bioinfiltration facilities have been designed and implemented to control flow, increase lag time, and infiltration capacity, and remove pollutants (Burns et al., 2012; Meals et al., 2010; Sage et al., 2015). Infiltration-based BMPs are the foundation of many sustainable infrastructure initiatives which focus on volume reduction near the source rather than “downstream” detention control (Emerson and Traver, 2008).

Despite increasing awareness of the ecological impact of stormwater on watershed function, the transition to sustainable land management that mimics natural hydrologic processes has been slow. This may be due to space limitations preventing installation of facilities for storing and infiltrating stormwater, inadequate knowledge for dealing with site-specific contaminant loads, and/or uncertainties about performance and cost (e.g., Roy et al., 2008). One challenge for implementation of volume-based infiltration practices is that runoff is cumulative when scaled up from site to watershed scale (Emerson and Traver, 2008). In highly altered landscapes, the receiving areas of these infiltration practices may not have adequate capacity to address runoff volumes from the contributing subwatershed. Although a watershed-wide runoff control design has yet to be developed, small scale and reach-scale strategies have shown promise for providing infiltration, intercepting sediment, and increasing pollutant removal before runoff reaches surface water (Zimmerman et al., 2010; Mayer et al., 2005; Hunt et al., 2008). However, without legislation to mandate the control and treatment of stormwater, there is less incentive for property owners to retrofit existing developments (Roy et al., 2008).

As cities become more densely populated and as farming operations become larger, the ability of the land surface to absorb water becomes increasingly limited. Significant climate change, population growth challenges, and societal expectations create a need for long-term and sustainable solutions to increase capacity for capturing and infiltrating surface water. Over the last few decades, several types of stormwater BMPs have been designed and implemented, but there have been limited efforts to monitor their performance after they have become established. The goal of my study was to examine some of these best management practices to determine their effectiveness for infiltration and pollutant removal, and to quantify their role in protecting surface water resources in both urban and agricultural landscapes.

Research Questions

Capture of stormwater runoff and pollutants by three types of urban best management practices

My first study was designed to determine the efficacy of three common vegetated stormwater best management practices (BMPs) in a variety of urban (residential, recreational, and commercial) environments. Specifically, I compared the spatial boundaries, physical characteristics, infiltration performance, absorption capacity, runoff response, and pollutant accumulation of each BMP receiving area to its subwatershed (contributing area). The practices I studied included bioretention cells, restored native landscaping (prairie), and riparian buffers. I explored the following questions: (1) Are receiving areas of these BMPs sized such that they provide adequate capacity for the capture of stormwater based on their original design criteria? (2) Do vegetated urban stormwater BMPs have greater infiltration rates and water absorption capacities than the surrounding contributing areas? and (3) Are there differences in pollutant (nutrient, heavy metal, and hydrocarbon) concentrations between the receiving and contributing areas of these practices?

Multi-species vegetated riparian buffers in rural and urban landscapes: Do they function similarly?

For my second study, I identified a single best management practice, forested riparian buffers, and I examined the similarities and differences in performance when implemented in urban and rural locations. Again, performance was measured by comparing characteristics of the buffers and their contributing subwatersheds. Performance was also analyzed to determine whether urban and rural buffers perform similarly compared to their respective contributing areas in terms of spatial boundaries, physical characteristics, infiltration, absorption capacity, and pollutant capture. Specifically, I explored the following questions: (1) Given the space limitations that are likely in urban landscapes, do urban riparian buffers have smaller practice area to contributing area ratios than rural buffers?(2) Are differences in infiltration rates and absorption capacity between buffer receiving areas and contributing landscape areas greater for urban buffers compared to rural buffers? and (3) Are there differences in the kinds and quantities of pollutants in runoff water and in soil for urban and rural buffers compared to their respective contributing areas?

Information gained from these studies could provide support for broader adoption of vegetated infiltration practices and contribute to better preparation for anticipated near and mid-term climate changes. Additionally, the results could indicate the degree to which stormwater management practices implemented under site-specific constraints can provide adequate peak flow management and pollution control.

Thesis Organization

The objectives of this thesis are addressed in the following sections: Chapter 1 is a general introduction; Chapter 2 is a manuscript entitled “Capture of stormwater runoff and pollutants by three types of urban best management practices;” Chapter 3 is a manuscript entitled “Multi-species vegetated riparian buffers in rural and urban landscapes: do they function similarly?;” and Chapter 4 is a general conclusion.

Author Contributions

The candidate was responsible for data collection, analysis, and the preparation of the text. Dr. Janette Thompson and Dr. Sally Logsdon provided guidance with experimental design, project execution, data collection, and manuscript editing. Dr. Logsdon was particularly instrumental in the soil analyses described in both Chapter 2 and Chapter 3. Dr. Peter Wolter provided comments and editorial recommendations on all chapters.

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CHAPTER 2: CAPTURE OF STORMWATER RUNOFF AND POLLUTANTS BY THREE TYPES OF URBAN BEST MANAGEMENT PRACTICES

A manuscript formatted for submission to the Journal of Soil and Water Conservation

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Abstract: Land cover changes associated with urbanization produce hydrological alterations which often diminish infiltration, leading to increased runoff volumes, peak flows, and greater need for pollution control. A number of urban best management practices (BMPs) have been designed to capture and contain stormwater runoff near the source. Although implementation of such practices has slowly increased, lack of evidence about their effectiveness in reducing the quantity and improving the quality of stormwater runoff may still limit the degree to which they are implemented. The objectives of this study were to assess performance of three types of urban stormwater BMPs by measuring their soil characteristics, infiltration rates, runoff reduction, and water quality parameters compared to adjacent contributing areas. Three types of practices - bioretention cells, native landscaping (restored prairie areas) and three-zone vegetated riparian buffers - located in Ames and Ankeny, IA were assessed by conducting infiltration tests and collecting soil and water samples. For the biocells in particular, practice surface areas were smaller in relation to their contributing areas than is recommended in current design criteria. On average, the bioretention cells and the buffers' wooded zones had significantly lower soil bulk densities, higher infiltration rates, and smaller runoff volumes than those of contributing areas. Time-to-runoff was particularly high for bioretention cells. Infiltration characteristics of the native landscapes (restored prairie) and buffer prairie zones we studied were not significantly different from those of the contributing areas. Total extractable hydrocarbon concentrations were elevated in the bioretention cells, while metals such as chromium had greater concentrations in the contributing areas. Based on these findings, we recommend careful attention to sizing, particularly for biocells, and suggest routine incorporation of soil amendments to improve the performance of restored prairie areas. Our findings also suggest that more widespread implementation of these source-control measures in retrofit of existing developments and/or in the design of newly urbanizing areas will be effective for reducing stormwater runoff volumes and improving water quality.

Keywords: stormwater management−bioretention cells−native landscaping−urban riparian buffers−infiltration rates−infiltrometer

Introduction

The process of urbanization increases the proportion of impervious surfaces in the landscape and generally leads to increased stormwater runoff. Both increased runoff volumes and higher peak discharge rates can disrupt natural drainage patterns and exceed the infiltration capacity of remaining pervious surfaces, leading to changes in the overall hydrologic flow regime (Booth and Jackson 1997). Rapid flow of stormwater across urban surfaces can deliver high concentrations of nutrients, metals, and hydrocarbons into nearby streams and lakes, leading to cumulative downstream impacts that damage surface water ecosystems (Beasley and Kneale 2002; Paul and Meyer 2001; Schueler 2000).

Historically, stormwater management was designed to attenuate flooding by removing water from the landscape quickly using “end of pipe” techniques in which impervious surfaces were directly connected to receiving waters via curbs, gutters, and storm drain pipes (Burns et al. 2012; Sage et al. 2015). Over time, a number of alternative practices have been developed aimed at retaining runoff water near the source and reduce pollutant loads before delivery to surface water systems (Prince George's County 1993; USEPA 2017). The enactment of Phase I (1990) and Phase II (2000) National Pollutant Discharge Elimination System (NPDES) rules created a legal mandate for many municipalities to incorporate structural management practices as one element in the "good housekeeping" permit requirement, although costs of doing so and lack of understanding limited early implementation of practices (Roy et al. 2008; USEPA 2000a). Currently, more is known about implementation of stormwater best management practices (BMPs), which can include cost-effective natural features that limit the quantity and treat the quality of stormwater runoff by capturing and processing it close to the area it is generated (Clar et al. 2004).

There is great potential for more use of these vegetated BMPs in both newly developing landscapes, such as Low Impact Development (LID) approaches (see for example Dietz 2007), and for retrofitting in areas of existing infrastructure (Sansalone et al. 2013). However, adoption and implementation of these practices in urban watersheds is not yet widespread, due to a variety of possible factors (e.g., lack of "proof" that they work, continued concerns about cost, inadequate guidelines for design and installation, inadequate governmental capacity and coordination, and/or lack of legal or economic incentives, as per Roy et al. 2008). There may also be real or perceived limitations related to space available for establishment of such practices (in the case of retrofitting) and urban residents' understanding of their role in the landscape (Page et al. 2015).

In this study, we examine the design criteria and efficacy of three vegetated stormwater management practices implemented in typical urban (residential, recreational and commercial) settings in central Iowa. Our goal is to address some of the uncertainties about BMP designs and performance so that they could be more widely adopted by both public and private entities, and ultimately reduce stormwater impacts. Specifically, we examine the characteristics of bioretention cells, native (restored prairie) landscaping, and riparian buffers, each of which is briefly described in the paragraphs that follow.

Bioretention cells are constructed depressions designed to infiltrate and temporarily store stormwater in order to decrease surface runoff, capture pollutants, and potentially recharge groundwater (Dietz 2007; LeFevre et al. 2015; Li and Davis 2009). Vegetation is included in these cells to promote evapotranspiration and maintain substrate porosity (Coffman et al. 1993). Design recommendations for biocells are regionally based, and vary in capacity, substrate media, and vegetation (Carpenter 2010). Often biocells are installed in parking lot islands, road medians, and in urban locations with relatively high and often immediately adjacent impervious cover.

Native landscaping is a cost-effective alternative to traditional turf grass that uses plant communities indigenous to a particular region to promote the natural hydrologic processing of runoff (Nassauer et al. 2009; Peterson et al. 2012). In the Midwest U.S., prairie plant communities have often been used in natural landscaping because seed and other propagules are readily available, restoration protocols are well understood, and the plant community is known to persist across a range of climatic conditions (e.g., thrive in both wet and dry years; Threlfall et al. 2017). The aboveground density of perennial prairie plants has been found to trap sediment and reduce surface runoff velocity (Ghadiri et al. 2011). Belowground, prairie plant root systems are thought to contribute to nutrient retention, stabilize soil structure, lower soil bulk density, and increase infiltration rates (Baer et al. 2002; Perez-Suarez et al. 2014).

Riparian buffers contain perennial vegetation planted along streambanks and the adjacent landscape, and are designed to slow water movement and prevent sediment and other pollutants from entering a stream while also providing for streambank stabilization (Laub et al. 2013; Parkyn et al. 2003; Polyakov et al. 2005; Roy at al. 2005). These often include species of trees and shrubs which are thought to increase water infiltration into the soil before it reaches the stream itself (Roy et al. 2005; Smucker and Detenbeck 2014; Wahl et al. 2013). In practice, vegetated riparian buffers implemented in urban landscapes have often been confined to relatively narrow parallel strips of land along stream corridors because of space limitations.

Our specific research objectives were to answer the following questions: (1) Are receiving areas of these BMPs sized such that they provide adequate capacity for the capture of stormwater based on their original design criteria? (2) Do vegetated urban stormwater BMPs have greater infiltration rates and water absorption capacities than the surrounding contributing areas? and (3) Are there differences in pollutant (nutrient, heavy metal, and hydrocarbon) concentrations between the receiving and contributing areas of these practices?

Materials and Methods

Study area

The landscape of Iowa has been extensively altered to accommodate both agricultural and urban land uses. As urban areas in Iowa have expanded, changes in land cover include conversion of former agricultural land as well as forest and grassland to residential, commercial, and industrial land uses dominated by impervious surfaces (Bowman et al. 2012). Although overall population growth has been slow in Iowa, it has been concentrated and occurred rapidly in several municipalities in the central part of the state, causing dramatic localized increases in road surfaces, parking lots, and buildings (e.g., a 19% increase in total impervious surface from 1940 to 2011 in Ankeny; Wu and Thompson 2013).

Several central Iowa cities are Municipal Separate Storm Sewer (MS-4) reporting entities permitted through EPA's National Pollutant Discharge Elimination System (NPDES) under Phase I and II rules, which require them to address a number of measures, including municipal "pollution prevention and good housekeeping." As a part of their permit responses to address this measure, a number of these cities have participated in the Iowa Stormwater Education Program, which provides technical assistance and support for municipal installation of stormwater BMPs. Among participating cities, we chose to assess practices installed within the City of Ames and the City of Ankeny, Iowa, incorporated boundaries (Figure 1).

Both cities are characterized by growing populations, which are estimated to have increased by 10% (Ames) and 25% (Ankeny) between 2010 and 2015. Current populations are estimated at 97,090 in Ames and 56,764 in Ankeny (US Census Bureau 2017). Population growth has led to rapid expansion of impervious surface coverage: recent studies indicate that the City of Ames has approximately 25% (city-wide average) impervious land cover (Jake Moore, personal communication) and the City of Ankeny has approximately 19% (in 2011) average impervious land cover (Wu and Thompson 2013).

We chose eight study sites within these two municipalities, including three bioretention cells, three native landscaping (restored prairie) areas, and two urban riparian buffers. These BMPs were chosen to represent structural practices embedded in catchment areas that include a variety of urban land cover types – roads, sidewalks, parking lots, and parks. All BMPs were installed five to fifteen years prior to this study and thus have had time for vegetative components to become established.

Study sites

Bioretention cells

The three bioretention cells selected for study include one in Ankeny (Summerbrook Park) and two in Ames (Edison Street and City Hall; Table 1). All three biocells are set approximately 23 cm (9 in) below the surrounding landscape, have a mulched and vegetated surface, a layer of engineered soil (between 30% and 60% sand, mixed with between 40% and 60% compost), and an aggregate rock layer enclosing a perforated drainage pipe. The Summerbrook Park biocell is located between a major road and a sidewalk. The two biocells in Ames both receive stormwater from municipal parking lots. All three bioretention cells were established in 2011 and are planted with mixtures of prairie grasses and forbs.

Native landscaping (Restored prairies)

The three restored prairies were selected to represent native landscaping, including two in Ames (in Ada Hayden Heritage Park and along Stange Road), and one in Ankeny (on the Iowa Association of Municipal Utilities grounds; Table 1). The Ada

Hayden prairie area is surrounded by a parking lot associated with the main access to this heavily used city park, and was established by direct seeding in 2007. The Iowa Association of Municipal Utilities (IAMU) prairie area is adjacent to the Carney Marsh Nature Preserve, and was first planted via direct seeding in 2001 (and re-seeded in 2006 to enhance the species mix) on land that was previously used for row crop agriculture (Dave Hraha, personal communication). Both the Ada Hayden and IAMU prairies are predominantly comprised of native prairie grasses with admixtures of some forbs. The Stange Road prairie area was established in 2004 by 4-H youth with support from the Iowa Department of Transportation's Living Roadway Trust Fund (Chris Strawhacker, personal communication). The soil at this site was amended with a compost mulch mixture just before planting. The Stange Road prairie was planted primarily with native forbs to increase its aesthetic appeal.

Urban riparian buffers

Two urban riparian buffers were selected and include a vegetated buffer planted in 2007 in Ames (Daley Park; described in Herringshaw et al. 2010) and a similar buffer planted in 2011 in Ankeny (Summerbrook Park). Each riparian buffer contains a zone of native tree species planted closest to the stream, with an intermediate zone of native trees mixed with shrubs, and furthest from the stream a zone of native prairie grasses and forbs. These linear zones of vegetation run parallel to the stream and perpendicular to nearby roadways, and vary in width throughout the reach of stream due to spatial constraints in each park. The Daley Park Buffer is located along a 310-m reach of College Creek in Ames, and the Summerbrook Park buffer is located along a 170-m stretch of an unnamed tributary in Ankeny. Both buffers are located in city parks located in residential neighborhoods.

Stormwater management practice area delineation

Practice areas and their contributing areas were delineated using a Geographic Information Systems (GIS) model. Light Detection and Ranging (LiDAR) data were used to generate digital elevation models (DEMs) at a one-meter resolution (Iowa LiDAR Mapping Project, GeoTREE 2017). These models were used to determine subwatershed boundaries and dimensions for the area draining to each of the stormwater management practice installations. Land cover data were extracted from the Natural Resources GIS Library in order to determine the area of impervious cover in each of the subwatersheds (NRGIS 2017).

Soil bulk density and volumetric water content measurements

Surface soil samples were collected adjacent to the location of three infiltrometer tests (described below) at each site to determine soil bulk density and soil water content. Soil samples were collected using an AMS Bulk Density Compact Slide Hammer™ to extract a 90.4-cm^3 cylindrical sample from the first 7 cm of soil. Samples were sealed in plastic bags, chilled immediately and transported in a cooler to the laboratory for further processing.

Soil bulk density samples were weighed, oven-dried at 70˚ C for 48 hours, and weighed again to determine mass of water and mass of soil in sample. Bulk density was calculated as soil dry weight divided by sample volume. Volumetric water content, expressed in $cm^3\ cm^{-3}$ was calculated using the following equation where

$$Volumetric\ water\ content\ \left(\frac{cm^3}{cm^3}\right) = \frac{m_{water}/\rho_{water}}{m_{soil}/\rho_{soil}} = \frac{\theta_g * \rho_{soil}}{\rho_{water}}$$

Where $\frac{m_{water}}{\rho_{water}}$ = the mass of the water in the soil over density of water, which is divided by $\frac{m_{soil}}{\rho_{soil}}$ = mass dry soil over the density of soil, or the gravimetric water content θ_g multiplied by bulk density over the density of water. Subsamples (40 g) were then mixed thoroughly with a sodium-hexametaphosphate solution (50 g L^{-1}) and analyzed to determine particle size distribution using the hydrometer method (ASTM 1985; Gee and Bauder 1986).

Infiltration measurements

A set of three infiltrometer tests were conducted in each practice area and in the surrounding contributing areas where possible (at three of the sites, those which were not dominated by pavement) between June and September, 2016. We used a portable single-ring Cornell Sprinkle Infiltrometer™ (van Es and Schindelbeck n.d.) to determine field saturated infiltration rates, time-to-runoff, and sorptivity. The cylindrical water reservoir of the infiltrometer has a perforated bottom which delivers rainfall onto a 24.1cm (9.5 inch) diameter area controlled by a 20.3 cm tall (8-inch) metal ring (van Es and Schindelbeck n.d.). Rainfall rate was calibrated by adjusting a Mariotte tube, which also controlled for constant head. Rainfall rate was calibrated each day to deliver 0.6 cm min^{-1} (0.24 in min^{-1}) using a two-minute test. Simulated rainfall rates were also determined directly to account for variation caused by field conditions (e.g., temperature variations). The reservoir was filled with deionized water transported from the laboratory, and the simulated rainfall rate was calculated by determining the difference in height of the water in the cylinder before and after the timed observation divided by the time elapsed.

A metal ring (17.8 cm height) fastened to the base of the reservoir was carefully inserted to be level with the soil surface to a depth of 7 cm. An outflow hole at 7 cm was set to be flush with the soil surface and was fitted with an effluent tube to allow water to run off once surface ponding occurred. We recorded rainfall rate, time-to-runoff, and volume of runoff water at three minute intervals. The runoff rate (cm min^{-1}) was calculated using the following equation (van Es and Schindlebeck n.d.):

$$runoff\ rate = \frac{V_t}{(457.30 * t)}$$

where 457.30 (cm²) = area of the metal ring, t = time interval (min), and V_t = volume of water (ml) during time interval t. After reaching a steady state for runoff rate, the field-saturated infiltration rate was estimated by determining the difference between the applied rainfall rate and the rate of runoff, allowing direct comparisons among sites with different antecedent soil moisture contents (van Es and Schindlebeck n.d).

Determination of practice capacity and potential water storage

Recommendations for practice design indicate that stormwater BMPs should be sized to capture/treat runoff from 90% of storms that occur in a typical year, which corresponds to a rainfall depth of 3.18 cm hr^{-1} (1.25-in hr^{-1}) in central Iowa (Iowa DNR 2009). However, given near-future climate scenarios that include more frequent intense rainfall events, we chose to estimate runoff generation for each site using a 5.08 cm hr^{-1} (2 in hr^{-1}) rainfall intensity. To estimate runoff for this hypothetical event, we subtracted the average infiltration rate (cm hr^{-1}) from this precipitation intensity and multiplied by the total surface area of each zone to determine the volume of water that would be generated (total runoff, indicated by a positive value) or infiltrated (total absorption, indicated by a negative value) by each BMP and contributing area.

Sample collection and analysis

Runoff water samples

All water samples were collected from the infiltrometer effluent tube at the time of initial runoff and were immediately chilled for transport and subsequent cold storage in the laboratory. Samples for measurement of nitrate concentration were collected in acid-washed (5% sulfuric acid; 2-hour rinse) and acidified (200 µg concentrated sulfuric acid) 125-ml bottles and analyzed using automated colorimetry (Method 353.2, USEPA 1993a). Acid-washed bottles treated in a phosphorous-free soap bath (2-hour rinse) were acidified and used to collect 125-ml samples for analysis of total phosphorus concentrations using USEPA semi-automated colorimetry (Method 365.1, USEPA 1993b). Acid-washed bottles were also used to collect 75-ml samples for determination of chloride concentrations using the low-level amperometric titration method (Method 4500-Cl E, APHA 2005). Samples were analyzed in the Riparian Management Systems Laboratory in the Department of Natural Resource Ecology and Management (nitrate) or in the Water Quality Research Laboratory in the Department of Agricultural and Biosystems Engineering (total phosphorus and chloride) at Iowa State University.

Surface soil samples

Soil samples were also collected from the soil surface to a 7-cm depth (approximately 3 in) at each site for determination of nutrient concentrations (nitrate, ammonium, and total phosphorus). These samples were placed in plastic-lined soil sample bags, chilled, and delivered to the Iowa State University Soil and Plant Analysis Laboratory where they were analyzed using KCl extraction and cadmium reduction detection methods for nitrate-nitrogen and ammonium, and the Mehlich-3 extraction and ascorbic acid spectrophotometric detection method for total phosphorus (NCRR 2015). Samples for determination of soil metal concentrations (chromium and zinc) were placed in 250-ml wide-mouth glass jars, chilled immediately, and delivered to the Iowa State Hygienic Laboratory within five hours of collection. Samples were analyzed there using inductively coupled plasma mass spectrometry (Method 6020A and Method 6010C, respectively; USEPA 2000b). Additional soil samples were collected at the three bioretention cells and restored prairie sites (all located close to motor vehicle traffic areas) for determination of total extractable hydrocarbons using the flame ionization capillary gas chromatography method developed by the State Hygienic Laboratory for extractable petroleum products (Method Iowa OA-2, UHL 1993). These samples were collected in 250-ml wide-mouth amber glass jars, chilled immediately, and delivered within five hours to the State Hygienic Laboratory for processing.

Soil cores from the Ames City Hall biocell

We used a hydraulic drilling rig with a plastic-lined tube to extract six soil cores to a depth of 51 cm (20 in). We divided each core into four, 13-cm (5-in) segments. A portion of each core segment was placed in a prepared container (as per previous samples) and immediately chilled for transport to the laboratory within five hours of collection. Analyses of nitrate, ammonium, and total phosphorus were conducted at the Soil and Plant Analysis Laboratory, Iowa State University, as per previously cited methods. Soil organic matter was estimated by determination of carbon through dry combustion, and soil pH was measured potentiometrically using an electronic pH meter in a one to one soil:water slurry at this laboratory (NCRR 2015). Metals (cadmium, chromium, and zinc) were analyzed at the State Hygienic Laboratory using the inductively coupled plasma mass spectrometry method for cadmium and chromium (Method 6020A, USEPA 1998) and the atomic emission spectrometry method for zinc (Method 6010C, USEPA 2000). Total extractable hydrocarbons and gasoline (as per previously cited methods) and *E. coli* using the multiple-tube fermentation technique (*Escherichia coli* procedure; Method 9221F, APHA 2005) were also measured at the State Hygienic Laboratory.

Data and statistical analyses

We calculated means for characteristics (soil physical and chemical properties, infiltration tests) of biocells and restored prairie areas using three samples from the three locations (means represent nine measurements, samples collected at random pattern). We calculated means for each buffer zone using three samples for each zone (prairie and wooded) at each site (means represent six measurements). We calculated means for contributing areas based on three samples from the three locations dominated by pervious surfaces surrounding one of the biocells and both urban riparian buffers (means represent nine measurements).

Comparison of practice means to contributing area means for soil bulk density and volumetric water content were made using student's t-tests. To account for possible correlation among multiple tests/samples from each site, we used a linear mixed-effects model fit by restricted maximum likelihood to estimate and compare means for infiltration characteristics (average infiltration rate, time-to-runoff, and runoff volume) for the practices and contributing areas (the LMER function in the R statistical package, Cook 2014). Estimates of time-to-runoff were converted to a log scale and were right-censored at 40 minutes (total test time) if 100% infiltration occurred. We used Student's t-tests for pairwise comparisons of means for water and soil chemical parameters for each practice type and the contributing areas. For detailed analyses of the City Hall biocell, means for soil physical and chemical characteristics were calculated using the six samples collected for each depth increment. We used Student's t-tests for pairwise comparisons among depth increments. For all statistical analyses, we set $p \leq 0.05$ to declare significance.

Results and Discussion

Characteristics of stormwater management practices and their contributing areas

Subwatershed contributing areas of the three bioretention cells had surface areas of 515 m^2 to 2060 m^2 (ratios from 1:14 to 1:28; Table 1). Bioretention cells were located in predominantly impervious landscapes – each of the three practices were surrounded by at least 46% impervious cover. The three restored prairie landscape areas received stormwater from contributing areas ranging from 2630 m^2 to 5478 m^2 (ratios of 1:0.5 to 1:2.3), including between 47% and 100% impervious cover. Subwatershed contributing surface areas surrounding the two riparian buffers were between 12,270 m^2 and 38,740 m^2, areas five to ten times larger than buffer zones themselves. These contributing areas were predominantly managed turf with some impervious cover (from 19% to 21%; Table 1).

Bioretention cell surfaces were much smaller than their surrounding contributing areas, which were characterized by relatively high proportions of impervious surfaces. According to Iowa Stormwater Management Manual (ISWMM 2016) guidelines, biocell surface areas should be approximately 5% to 10% of the contributing surface area. The bioretention cells we examined had surface areas that were somewhat low compared to this recommendation, representing between 3.5% and 7% of the surrounding subwatershed, but had practice area to contributing area ratios similar to those reported in other studies (e.g., Houdeshel and Pomeroy 2014; Johnson and Hunt 2016). Although we studied only one biocell at each site, each of them were co-located with other biocells that likely increase overall capacity for source control treatment in these landscapes.

The three native landscaping (restored prairie) areas had much larger surfaces relative to their contributing areas. Native landscaping is increasingly recommended for use in urban areas (Fischer et al. 2013; Reid and Oki 2008), although space available or urban dwellers' aesthetic preferences (Borgstrom et al. 2006; Lerman et al. 2012; Peterson et al. 2012) in many urban settings may limit its potential for application on private property. Native prairie landscaping in particular may be most appropriate at large scales, focusing on municipal or commercial properties, and using design plans that include specific maintenance methods and schedules.

The riparian buffers we examined had intermediate surface area to contributing area ratios (1:4 for combined prairie and wooded zones within each buffer) compared to the other two practices. Because of their purpose and landscape position as a linear feature along stream corridors, recommendations generally address buffer width rather than surface area - in Iowa, recommended width ranges from 4.5 to 7.6 m (15 to 25 ft., ISWMM 2016). The riparian buffers we observed had variable widths ranging from 10 to 40 m, exceeding suggested design criteria. Similar to applications of native landscaping, the total area available for a riparian buffer may be quite constrained in urban settings, thus other reports of urban buffer widths vary greatly (from 5 to 60 m; see for example Johnson and Buffler 2008; Schueler 1995).

Soil physical properties and practice infiltration characteristics

Surface soil cores from the three bioretention cells had lower soil bulk density than their respective contributing areas ($p = 0.0117$; Table 2). Soil bulk density in the restored prairie areas and in the buffer prairie zones did not differ from their contributing areas ($p = 0.6626$ and 0.7610, respectively). The buffer wooded zones had lower soil bulk density than their respective contributing areas ($p = 0.0285$). Volumetric soil water content was not significantly different for bioretention cells, prairie landscape areas or buffer prairie zones compared to their contributing areas, but for buffer wooded zones it was greater than that of their respective contributing areas ($p = 0.0301$; Table 2).

Average infiltration rates were greater for the bioretention cells ($p < 0.0001$) and buffer wooded zones ($p < 0.0001$), compared to their respective contributing areas (Table 3). Average infiltration rates for the restored prairie landscape areas and the buffer prairie zones were not significantly different from their contributing areas. Bioretention cells were also characterized by longer time-to-runoff compared to their respective contributing areas ($p < 0.0001$). There were no consistent differences in time-to-runoff for the other practices. Both bioretention cells ($p = 0.0002$) and buffer wooded zones ($p = 0.0004$) produced smaller volumes of runoff than their respective contributing areas, although there were no differences between restored prairie landscapes and buffer prairie zones compared to their contributing areas (also Table 3).

Infiltration rates were consistently high in practice areas, and were significantly greater for biocells and buffer wooded zones compared to the contributing areas. This is probably related to lower soil bulk density for substrate materials in these two practices leading to more pore space for water infiltration. The bioretention cells were specifically created to have low bulk density as per recommended guidelines using engineered soil mixtures (on average surface samples were 73% sand, ranging from sand to sandy loam textures based on particle size analysis, data not shown). We also determined that soils in the buffer wooded zones ranged from sandy loam to loam textures with an average 45% sand content, likely enhancing their permeability.

We observed high variability in infiltration rates among the three restored prairie landscape areas. Although thought to be a better alternative than managed turf lawns, several previous studies have revealed the potential for variable effects of prairie that are relevant to its potential for stormwater management. For example, Gish and Jury (1983) found that prairie plant roots created soil physical conditions leading to a narrow range of pore water velocities that actually reduced infiltration rates.

The somewhat high soil bulk densities and low infiltration rates we observed could be related to those factors, or to conditions that existed or were created at the time of prairie establishment. For example, Ada Hayden Heritage Park is located at the site of a former gravel quarry, where initial soil bulk density may have been very high at the time of prairie installation (Joshua Thompson, personal communication). The prairie at the Iowa Association of Municipal Utilities facility was planted on land previously used for row crop production, so soil physical properties at this site were also likely to have been altered by prior land use. The same may be true for the prairie zones associated with the two buffers, which were planted in urban landscapes that had been graded. In such situations, pre-treatment of the area with soil amendments such as compost may be necessary to enhance soil properties before seeding to prairie (see for example, Singer et al. 2006) to achieve desired infiltration characteristics.

The buffer wooded zones that we sampled were characterized by relatively low soil bulk density values and generally high infiltration rates. This may be attributable to their position closest to the stream in areas less likely to have been disturbed by prior landscape alterations, in addition to the role of woody vegetation in creating large pores that increase water movement (Dexter 1991). Others have found that soil bulk density in restored urban vegetated riparian buffers was intermediate between that of urban control sites (no treatment) and naturally forested streamside areas (Laub et al. 2013). We also measured significantly higher volumetric water content of soils in buffer wooded zones, which could be due to greater infiltration capacity as well as contributions from subsurface base flow based on their topographic position and proximity to the stream (Bosch et al. 1994; Sweeney and Newbold 2014).

Time-to-runoff was consistently high and runoff volume was consistently low for practice areas, although we detected significant differences only for the bioretention cells compared to contributing areas. The fact that we did not observe distinguishable responses in time-to-runoff for some practice areas relative to the contributing areas may be due to the more important role of rainfall intensity as a determinant of time-to-runoff (Bothma et al. 2012). The rainfall rate we used during infiltrometer tests (0.6 cm min^{-1}, or approximately 14 in hr^{-1}) may have uniformly caused surface ponding or slaking of soil aggregates regardless of substrate properties, affecting both practice areas and contributing areas. The lack of significant differences for practice areas compared to contributing areas could also be due to the high amount of variability in these parameters within the practices themselves and limitations on the number of tests we were able to conduct.

Practice capacity and potential water storage

One of the three restored prairie areas and both riparian buffers including each of the buffer (wooded and prairie) zones had adequate capacity to infiltrate directly incident precipitation from a 5.08 cm hr^{-1} (2 in hr^{-1}) event, as well as all of the runoff generated from their contributing areas (Table 4). The three bioretention cells were unable to absorb the quantity of runoff generated by their surrounding contributing areas for this hypothetical event; for instance, the City Hall bioretention cell was estimated to absorb only 6.6 m^3 hr^{-1} of the 26.2 m^3 hr^{-1} runoff generated. One restored prairie area was estimated to absorb even less than the direct incident precipitation that would be delivered to it by a rain event of this intensity (Table 4).

Designed practice depths for the three bioretention cells ranged from 61 cm to 91 cm (Table 5). Potential water storage depths in the bioretention cells were estimated to range from 11 cm to 30 cm. A uniform practice depth of 91 cm (3 ft) was assigned for the restored prairie landscapes. Depths of potential water storage for the restored prairie areas ranged from 18 cm to 34 cm (Table 5). Estimates of potential water storage for the buffer prairie zones ranged from 21 cm to 28 cm, and for the buffer wooded zones ranged from 26 cm to 44 cm (Table 5).

Guidelines for stormwater control practices for Iowa are based on design criteria of 3.2 cm (1.25 in) of rainfall, which historically represented 90% of such events (Iowa SUDAS 2015; ISWMM 2016). However, based on current climate change scenarios, more frequent and intense rainfall events are very likely (e.g., Takle et al. 2010; Wu et al. 2013). Therefore, we tested whether these practices had the potential to mitigate runoff from a 5 cm (2 in) rainfall event, representing a one-hour storm previously estimated to have a return period between 5 to 10 years (Iowa SUDAS 2015). For this storm intensity, the individual bioretention cells we studied are undersized for the quantity of stormwater production from their contributing areas (which include a predominance of impervious surface). Thus, it may be advisable to adjust design criteria to accommodate increased intensity of anticipated rainfall events. In addition, this underscores the necessity for establishing such practices in clusters to increase their effectiveness (as recommended, but not always complied with for practice installations). Restored prairie areas generated runoff quantities similar to those of their surrounding contributing landscapes, again suggesting that soil amendments before prairie establishment (Singer et al. 2006) could enhance their performance.

Wooded zones of the riparian buffers we studied absorbed more stormwater runoff than their adjacent prairie zones. Suspended sediment has been observed to settle within the first 10 to 15 feet of vegetated areas meant to intercept and treat runoff (Hunt and Lord 2006). The location of the prairie buffer zone at the outer edge of both vegetated buffers may lead to sediment accumulation in these prairie zones, which could fill pores, increase their bulk density, and decrease their infiltration capacity.

Water sample characteristics

There were no differences in nitrate concentrations of effluent runoff water for any of the practice areas compared to their contributing areas. Total phosphorus concentrations in runoff were significantly lower for all practices (p-values ranging from 0.0013 to 0.0532) compared to those of the contributing areas (Table 6). Chloride concentrations in runoff water were lower for restored prairie areas ($p = 0.0129$) and buffer wooded zones ($p = 0.0443$) compared to contributing areas (also Table 6).

Phosphorus and nitrogen are primary nutrient pollutants found in stormwater runoff (USEPA 2009). Relatively low concentrations of nitrate, total phosphorus, and chloride in the effluent water samples we collected may have been a result of the short travel time/distance across the soil surface in our tests (maximum travel distance was 24 cm, the diameter of the infiltration ring). Consideration of nitrates in runoff water is essential for managing stormwater quality, but surface runoff is generally not seen as a dominant pathway for nitrate transport (Kleinman et al. 2006). Phosphorus concentrations in effluent water were significantly lower in all four practice types than the contributing areas, inconsistent with the phosphorus concentrations found in the soils of the practices. Although dissolved P concentration in surface runoff is likely to be related to soil P concentrations, some studies have observed that the relationship between soil and runoff phosphorus content depends on several site-specific factors (e.g., Kleinman et al. 2006, Nash et al. 2002; Sharpley et al. 1994) which we did not measure.

Higher chloride concentrations in contributing areas and somewhat elevated chloride concentrations in effluent runoff from bioretention cells and buffer prairie zones are probably due to residue from road salts used on adjacent impervious surfaces. These practice areas likely receive inputs of sodium chloride and calcium chloride that are used to treat snow and ice (Zhang et al. 2013). The restored prairie areas and buffer wooded zones had lower chloride concentrations in their effluent water, likely because they are in landscape positions that are protected from salt inputs.

Soil sample characteristics

We did not detect differences for soil nitrate, ammonium, total phosphorus, zinc or total extractable hydrocarbons in surface soil samples in practice areas compared to contributing areas (Table 7). Soil chromium concentrations were significantly lower in bioretention cells than in contributing areas ($p = 0.0201$; Table 7). Although levels of extractable hydrocarbons are elevated in bioretention cells, variability within practice areas precluded detection of a significant difference between these practices and the contributing landscapes.

We did not detect differences in soil nutrient, metal, or hydrocarbon concentrations between the practices and contributing areas, which could be because there was high variability within each practice for the soil parameters we measured. For example, total extractable hydrocarbon concentrations were around 30 mg kg^{-1} for two of the bioretention cells (Summerbrook Park and Edison Street) but the third biocell (City Hall) had a concentration of 200 mg kg^{-1}. Although design guidelines that target specific pollutants have not been developed within the ISWMM standards, a target infiltration rate for capture of metals, total nitrogen, and total phosphorus between 2.54 to 15.24 cm hr^{-1} (1 to 6 in hr^{-1}) for adequate soil absorption has been suggested (Hunt and Lord, 2006). It is possible that the relatively low total phosphorus and chromium soil concentrations measured in the biocells and buffer wooded zones were due to their high infiltration rates. Although a number of researchers have documented removal rates for pollutants in stormwater control practices (e.g., Hatt et al. 2009; Wilkins et al. 2015) few have examined nutrient and metal concentrations retained within the practice substrates themselves. The data from our study is therefore helpful for understanding the capacity of these practices for pollutant retention/storage.

Detailed analysis of City Hall bioretention cell

Mean nitrate concentrations were significantly greater in the surface two increments of the soil core samples ($p < 0.0001$) and decreased with depth (Table 8). Ammonium and total phosphorus concentrations were significantly lower in the surface two core increments and increased with depth. Percent organic matter was significantly greater in the surface core increment. Cadmium and chromium concentrations did not vary with depth. Zinc concentrations in the surface increment of soil cores were significantly higher than for deeper core increments ($p = 0.0001$). Soil pH was significantly lower at the surface compared to the other depth increments (Table 8).

We were surprised to find that ammonium concentrations increased with soil depth and nitrate concentrations decreased with depth in the City Hall bioretention cell. Typically ammonium is oxidized through the nitrification process under aerobic conditions to form nitrate (Rittman and McCarthy 2001). However, anoxic conditions created by soil saturation would prevent nitrification, resulting in high retention of both ammonium and nitrate (Baker and Vervier 2004; Dietz and Clausen 2006; Forshay and Stanley 2005). Dissimilatory nitrate reduction to ammonium is another mechanism that could explain the increase of ammonium in cores from greater depths in this biocell (Sgouridis et al. 2011). Lastly, higher concentrations of ammonium with depth could simply be a result of leaching due to high sand content (55.5%) and high infiltration rates. Sandy soils, which have low ionic sorption capacities and provide more pore space for water percolation would speed up movement of nutrients and limit opportunities for retention and more typical chemical transformations (McPharlin et al. 1994; Pathan et al. 2002).

Total phosphorus concentrations increased with depth in this biocell. This may be related to the substrate mixture (60% compost) releasing P that then accumulates at depth. For example, Paus et al. (2014) found that P was released at 203 ± 24 mg P kg^{-1} of soil media in the compost column of bioretention cells they studied. Other researchers have also found that bioretention cell soil media with high concentrations of organic matter can release both organic and inorganic P during decomposition, which could be transported to greater depths (e.g., Hatt et al. 2009; LeFevre et al. 2015).

Concentrations of cadmium and chromium were generally low and did not vary with core increment depth. In previous research, cadmium has been shown to accumulate in the surface layers of bioretention cells, thus the low and consistent concentrations we observed probably indicate low input from the surrounding contributing areas (Udom et al. 2004; Wang et al. 2016). Zinc concentrations were considerably higher in the surface core increments we tested, indicating contributions from the surrounding parking lot (due to residue from rubber tires, vehicle exhaust and motor oil additives) and high adsorption/low mobility of this metal. Other studies have also shown that bioretention cells can effectively immobilize zinc from stormwater runoff (Davis et al. 2003; Li and Davis 2009). Levels of total extractable hydrocarbons, gasoline and *E. coli* were also significantly greater at the surface and did not accumulate at greater depths. This suggests the bioretention cell is effectively retaining these pollutants, although further investigations of inflow and outflow water would be necessary to verify that function (Chapman and Horner 2010).

Summary and Conclusions

Uncertainties about performance, design guidelines, lack of public acceptance, and limitations on space available to install practices that control runoff generated in urban settings can be impediments to stormwater BMP implementation. However, all practices observed in this study were characterized by relatively high infiltration rates and demonstrated capacity to contain water and pollutants compared to contributing areas. Further, these practices were successfully retrofitted into a variety of existing land uses under the purview of either municipal governments or a commercial entity, indicating that the application of BMPs does not have to be limited by urban land use or space constraints.

The physical properties of the substrates used in the bioretention cells we examined contributed to high infiltration rates, longer time-to-runoff and greater pollutant accumulation compared to contributing areas. Additional design adaptations could include expansion of practice surface areas to enhance their performance during anticipated frequent intense storm events, and by customization of substrate amendments for removal, absorption, and transformation of the specific pollutants expected in the landscapes where biocells are to be installed. For greatest effectiveness, biocell installation should include placement of curb cuts for street-side stormwater entry, as well as installation of forebays at the point(s) of entry to capture sediment and prevent surface clogging.

Benefits of native (restored prairie) landscaping include ease of integration in a variety of urban settings, reduced need for maintenance (e.g., regular irrigation and/or mowing), and creation of habitat that could support other forms of native biodiversity.

Based on the sites we studied, application of soil amendments prior to establishment of native plants is probably necessary to increase infiltration rates and capacities of these features by decreasing bulk density and through increasing plant density (Singer et al. 2006).

The three-zone buffers observed in this study provided surfaces which stormwater could flow over or through. Buffer wooded zones closest to the streambanks performed very well for infiltration and absorption. Further, although full-stream-length buffers are known to be most effective, urban riparian buffers (including those observed in this study) function adequately even when implemented on a more limited reach-scale to accommodate existing infrastructure or fit within space under direct municipal management (e.g., public parks).

We determined that even though most of the BMPs we assessed are somewhat undersized, they do have greater infiltration rates and absorption capacities than their surrounding contributing areas and likely provide adequate source control for frequent low-intensity rain events. We observed some pollutant accumulation in the BMPs we studied, and suggest that future research could more intensively investigate this aspect of BMP performance. The conservation implications of this study are that vegetated source-control best management practices are effective, that increased implementation of these practices is warranted, and that modifications to design criteria for such practices could provide additional protection for surface water systems (streams, rivers and lakes) against peak runoff flows and the pollutants they often carry in urban areas, particularly under predicted future climate scenarios.

Acknowledgements

Support for this study was provided by the Iowa Department of Agriculture and Land Stewardship (State Soil Conservation Committee), the Center for Global and Regional Environmental Research, the Department of Natural Resource Ecology and Management at Iowa State University, the National Laboratory for Agriculture and the Environment, and McIntire-Stennis funds. We thank the City of Ames, City of Ankeny, and the Iowa Association of Municipal Utilities for allowing access to sites to conduct tests and collect samples. We also thank Gavin Simmons, Bret Lang, Sarah Rueger, Matthew Kessler, Bree Marmur, and Morgan Davis for field and laboratory assistance. We appreciate statistical advice from Eric Mittman, and additional advice and information provided by Gavin Simmons, Pat Sauer, Leigh Ann Long, Dr. Peter Wolter, Amy Bryant, Jake Moore, Dave Hraha, and Joshua Thompson.

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Table 1: General characteristics of sites where three stormwater management practices were evaluated in central Iowa, USA. Surface areas of management practices on eight sites and their respective contributing areas (CA) by practice type, ratios of practice area to contributing area, and total impervious surface cover (roads, other pavement, roofs; by area and percent) within each contributing area.

BMP type/site	Location	Surrounding Land use	Year	Surface area, m²	Practice area to CA ratio	Impervious cover, m² (%)
Bioretention cells						
Summerbrook Park	Ankeny	City park	2011	35	1 : 28	
Contributing area				1006		462 (46%)
Edison Street	Ames	Municipal parking lot	2011	143	1 : 14	
Contributing area				2060		1520 (74%)
City Hall parking lot	Ames	Municipal parking lot	2011	37	1 : 14	
Contributing area				515		458 (89%)
Restored prairie areas						
Ada Hayden Heritage Park	Ames	City park	2007	3279	1 : 1	
Contributing area				3145		3145 (100%)
Iowa Assoc. Municipal Utilities	Ankeny	Commercial area	2001	10151	1 : 0.5	
Contributing area				5478		2592 (47%)
Stange Road	Ames	University grounds	2004	1118	1 : 2	
Contributing area				2630		1757 (67%)
Urban riparian buffers						
Daley Park buffer prairie				4200	1 : 9	
Daley Park buffer wooded	Ames	City park	2007	8070	1 : 5	
Contributing area				38740		8236 (21%)
Summerbrook Park buffer prairie				1490	1 : 10	
Summerbrook Park buffer wooded	Ankeny	City park	2011	2400	1 : 6	
Contributing area				12270		2349 (19%)

Table 2: Soil bulk density (g cm^{-3}) and volumetric water content (cm^3 cm^{-3}) for stormwater practices and their contributing areas (managed turf). Means for the same set of three contributing areas are used for comparison to all practices. Estimated mean differences between practices and the contributing areas, their standard errors, and p-values for comparisons using Student's t-tests.

		Soil bulk density (g cm^{-3})			Volumetric water content (cm^3 cm^{-3})		
Practice	n	Mean	Estimate of difference (Std. error)	p-value	Mean	Estimate of difference (Std. error)	p-value
Bioretention cells	9	1.16	0.24 (0.09)	0.0117	0.25	0.01 (0.05)	0.8218
Restored prairie areas	9	1.35	0.04 (0.09)	0.6626	0.27	0.01 (0.05)	0.8255
Buffer prairie zones	6	1.36	0.03 (0.10)	0.7610	0.27	0.01 (0.05)	0.8936
Buffer wooded zones	6	1.18	0.23 (0.10)	0.0285	0.38	0.11 (0.05)	0.0301
Contributing areas	9	1.40			0.26		

Table 3: Means and estimates of average differences between stormwater management practices and contributing areas (managed turf). Means for the same set of three contributing areas are used for comparison to all practices. Parameters measured include average infiltration rate (cm hr^{-1}), time-to-runoff (minutes, right-censored), and runoff volume (liters).

Infiltration characteristic	n	Mean	Estimate of difference	Std. error	df	t ratio	p-value
Average infiltration rate, cm hr^{-1}							
Bioretention cell	9	33.79	31.77	4.56	27.84	6.96	< 0.0001
Restored prairie	9	15.54	13.52	6.58	4.61	2.05	0.1001
Buffer prairie zone	6	6.94	4.91	3.66	26.51	1.34	0.1914
Buffer wooded zone	6	23.00	20.97	4.11	30.13	5.11	< 0.0001
Contributing areas	9	2.03					
Time-to-runoff, minutes							
Bioretention cell	9	3.10	3.35	1.63	23.9	18.16	<0.0001
Restored prairie	9	2.12	2.37	9.13	4.9	1.81	0.1302
Buffer prairie zone	6	1.15	1.40	1.26	23.1	1.78	0.0888
Buffer wooded zone	6	0.95	1.21	1.34	23.4	1.08	0.2901
Contributing areas	9	-0.25					
Runoff volume, liters							
Bioretention cell	9	2.13	-7.23	1.55	18.3	-4.67	0.0002
Restored prairie	9	4.93	-4.43	1.62	3.3	-2.74	0.0650
Buffer prairie zone	6	7.11	-2.25	1.36	26.3	-1.65	0.1114
Buffer wooded zone	6	3.13	-6.23	1.56	31.9	-3.40	0.0004
Contributing areas	9	9.36					

Table 4: Determination of absorption capacity for each of three stormwater management practices and their contributing areas assessed at eight sites in central Iowa, based on average infiltrations rates and surface areas.

BMP type/site	Average infiltration rate, cm hr^{-1}	Surface area, m^2	Impervious cover, m^2 (%)	Runoff generation, m^3 hr^{-1}	Total absorption, m^3 hr^{-1}
Bioretention cells					
Summerbrook Park	39.5	35			-12.0
Contributing area	1.9	1006	462 (46%)	33.2	
Edison Street	39.9	143			-49.8
Contributing area	-	2060	1520 (74%)	104.6	
City Hall parking lot	22.2	37			-6.3
Contributing area	-	515	458 (89%)	26.2	
Restored prairie areas					
Ada Hayden Heritage Park	27.4	3279			-731.3
Contributing area	-	3145	3145 (100%)	925.2	
Iowa Assoc. Municipal Utilities	3.9	10151		121.2	
Contributing area	-	5478	2592 (47%)	278.3	
Stange Road	17.2	1118			-135.5
Contributing area	-	2630	1757 (67%)	133.6	
Urban riparian buffers					
Daley Park prairie zone	6.2	4198			-45.9
Daley Park wooded zone	29.9	6149			-1530.6
Contributing area	5.7	38739	8236 (21%)		-236.2
Summerbrook Park prairie zone	21.3	1221			-198.2
Summerbrook Park wooded zone	9.9	1871			-90.8
Contributing area	4.3	12270	2349 (19%)	90.4	

Table 5: Stormwater management practice water holding capacities prior to sampling based on practice dimensions (biocells) or estimated root depth in soil (prairie landscape areas and buffer prairie and wooded zones).

BMP type/site	Volumetric water content, $cm^3\ cm^{-3}$	Practice depth, cm	Depth of water held, cm	Practice volume, m^3	Volume of water held, m^3
Bioretention cells					
Summerbrook Park	0.15	76.20	11.51	26.52	4.00
Edison Street	0.29	60.96	17.43	87.17	24.93
City Hall parking lot	0.33	91.44	29.72	33.80	10.99
Restored prairie areas					
Ada Hayden Heritage Park	0.20	91.44	18.20	2998.32	596.67
Iowa Assoc. Municipal Utilities	0.38	91.44	34.29	9282.07	3480.77
Stange Road	0.25	91.44	22.95	1022.40	256.62
Urban Riparian Buffers					
Daley Park prairie zone	0.24	91.44	21.21	3352.50	820.69
Daley Park wooded zone	0.30	91.44	25.97	1949.88	581.84
Summerbrook Park prairie zone	0.33	91.44	28.44	3202.02	1065.95
Summerbrook Park wooded zone	0.53	91.44	43.89	2862.63	1506.88

Table 6: Means for each stormwater management practice type for effluent runoff water nitrate, total phosphorus, and chloride concentrations. Means for the same set of three contributing areas are used for comparison to all practices using student's t-tests.

Effluent runoff concentration	n	Means	Difference of means	Std. error	p-value
Nitrate, mg kg^{-1}					
Bioretention cell	3	0.10	0.08	0.10	0.4534
Restored prairie	8	0.14	0.04	0.07	0.5955
Buffer prairie zone	5	0.15	0.03	0.08	0.7156
Buffer wooded zone	6	0.28	0.10	0.08	0.2106
Contributing areas	9	0.18			
Total Phosphorus, mg kg^{-1}					
Bioretention cell	3	0.34	0.28	0.14	0.0532
Restored prairie	8	0.25	0.36	0.10	0.0013
Buffer prairie zone	5	0.34	0.27	0.11	0.0267
Buffer wooded zone	6	0.30	0.31	0.11	0.0078
Contributing areas	9	0.61			
Chloride, mg kg^{-1}					
Bioretention cell	3	0.96	1.49	0.89	0.1081
Restored prairie	8	0.70	1.74	0.65	0.0129
Buffer prairie zone	5	1.66	0.79	0.75	0.3012
Buffer wooded zone	6	0.95	1.49	0.71	0.0443
Contributing areas	9	2.45			

Table 7: Means and difference of means for surface soil sample concentrations of nitrate, ammonium, total phosphorus, zinc, chromium, and total extractable hydrocarbons. P-values for comparisons between stormwater practices and their contributing areas are based on pairwise student's t-tests.

Soil nutrient or metal concentrations	n	Mean	Difference of means	Std. error	p-value
Nitrate, mg kg^{-1}					
Bioretention cells	9	7.2	4.5	3.2	0.1950
Restored prairies	9	1.3	1.5	3.2	0.6535
Buffer prairie zones	6	7.2	4.4	3.5	0.2431
Buffer wooded zones	6	2.1	0.7	3.5	0.8436
Contributing areas	9	2.8			
Ammonium, mg kg^{-1}					
Bioretention cells	9	2.3	2.2	1.1	0.0948
Restored prairies	9	3.5	0.9	1.1	0.4185
Buffer prairie zones	6	4.5	0.0	1.3	0.9797
Buffer wooded zones	6	4.4	0.1	1.3	0.9289
Contributing areas	9	4.5			
Total phosphorus, mg kg^{-1}					
Bioretention cells	9	88.0	17.3	33.8	0.6224
Restored prairies	9	91.7	21.0	33.8	0.5522
Buffer prairie zones	6	100.5	29.8	37.8	0.4532
Buffer wooded zones	6	49.5	21.2	37.8	0.5912
Contributing areas	9	70.7			
Zinc, mg kg^{-1}					
Bioretention cells	9	39.7	1.3	31.6	0.9673
Restored prairies	9	62.3	21.3	31.6	0.5180
Buffer prairie zones	6	47.5	6.5	35.3	0.8584
Buffer wooded zones	6	110.5	69.5	35.3	0.0843
Contributing areas	9	41.0			
Chromium, mg kg^{-1}					
Bioretention cells	9	7.1	8.2	2.8	0.0201
Restored prairies	9	19.0	3.6	2.8	0.2317
Buffer prairie zones	6	14.5	0.8	3.2	0.7991
Buffer wooded zones	6	9.3	6.0	3.2	0.0933
Contributing areas	9	15.3			
Total extractable hydrocarbons, mg kg^{-1}					
Bioretention cells	9	88.0	30.0	82.2	0.7337
Restored prairies	9	41.7	16.3	82.2	0.8522
Contributing areas	9	58.0			

Table 8: Detailed analysis of City Hall bioretention cell. Six soil cores were extracted to a depth of 50.8 cm (20 in) and divided into four, 12.7 cm (5 in) segments. Depth increments are labeled as follows: A = 0 - 12.7 cm (0 - 5 in), B = 12.7 - 25.4 cm (5 - 10 in), C = 25.4 - 38.1 cm (10 - 15 in), and D = 38.1 - 50.8 cm (15 - 20 in). P-values are based on comparisons using student's t-tests.

Depth Increment	Mean	Mean difference (Std. error)	p-value	Mean	Mean difference (Std. error)	p-value	Mean	Mean difference (Std. error)	p-value	Mean	Mean difference (Std. error)	p-value
	Nitrate, mg kg^{-1}			Ammonium, mg kg^{-1}			Total phosphorus, mg kg^{-1}			Organic matter, %		
A	3.35	2.12	0.0001	1.85	0.98	0.6308	114.7	5.67	0.7244	7.23	3.63	0.0001
B	1.23	(0.39)		2.83	(2.02)		120.3	(15.85)		3.55	(0.69)	
A	3.35	2.58	0.0001	1.85	7.92	0.0008	114.7	15.50	0.3397	7.23	4.35	0.0001
C	0.76	(0.39)		9.77	(2.02)		130.2	(15.85)		2.88	(0.69)	
A	3.35	3.05	0.0001	1.85	14.0	0.0001	114.7	44.16	0.0114	7.23	3.80	0.0001
D	0.30	(0.39)		15.85	(2.02)		158.8	(15.85)		3.43	(0.69)	
B	1.23	0.46	0.2498	2.83	6.93	0.0026	120.3	9.83	0.5419	3.55	0.67	0.3485
C	0.76	(0.39)		9.77	(2.015)		130.2	(15.85)		2.88	(0.69)	
B	1.23	0.93	0.0279	2.83	13.02	0.0001	120.3	38.5	0.0247	3.55	0.12	0.8683
D	0.30	(0.39)		15.85	(2.02)		158.8	(15.85)		3.43	(0.69)	
C	0.76	0.46	0.2498	9.77	6.08	0.0068	130.2	28.67	0.0855	2.88	0.55	0.4376
D	0.30	(0.39)		15.85	(2.02)		158.8	(15.85)		3.43	(0.69)	
	Cadmium, mg kg^{-1}			Chromium, mg kg^{-1}			Zinc, mg kg^{-1}			pH		
A	2.02	0.02	0.1727	6.30	0.13	0.7969	65.33	34.50	0.0001	8.08	0.22	0.0449
B	2.00	(0.01)		6.16	(0.51)		30.83	(4.26)		8.31	(0.11)	
A	2.02	0.02	0.1727	6.30	0.32	0.5426	65.33	35.00	0.0001	8.08	0.22	0.0527
C	2.00	(0.01)		6.62	(0.51)		30.33	(4.26)		8.30	(0.11)	
A	2.02	0.02	0.1727	6.30	0.23	0.6529	65.33	35.66	0.0001	8.08	0.30	0.0098
D	2.00	(0.01)		6.53	(0.51)		29.66	(4.26)		8.38	(0.11)	
B	2.00	0.0	1.0	6.16	0.45	0.3891	30.83	0.50	0.9077	8.31	0.01	0.9376
C	2.00	(0.01)		6.62	(0.51)		30.33	(4.26)		8.30	(0.11)	
B	2.00	0.0	1.0	6.16	0.37	0.4814	30.83	1.17	0.7870	8.31	0.08	0.4840
D	2.00	(0.01)		6.53	(0.51)		29.66	(4.26)		8.38	(0.11)	
C	2.00	0.0	1.0	6.62	0.08	0.8721	30.33	0.67	0.8772	8.30	0.08	0.4375
D	2.00	(0.01)		6.53	(0.51)		29.66	(4.26)		8.38	(0.11)	

Table 8: Continued.

Depth Increment	Mean	Mean difference (Std. error)	p value	Mean	Mean difference (Std. error)	p value	Mean	Mean difference (Std. error)	p value
	Total Extractable Hydrocarbons			Gasoline, mg kg^{-1}			E. coli, mg kg^{-1}		
A	200	171.8	0.0040	45.5	42.5	0.0001	30.4	27.3	0.0030
B	28.2	(52.8)		3.0	(8.8)		3.1	(8.1)	
A	200	185.2	0.0022	45.5	42.5	0.0001	30.4	27.3	0.0031
C	14.8	(52.8)		3.0	(8.8)		3.1	(8.1)	
A	200	110.0	0.0504	45.5	42.5	0.0001	30.4	27.4	0.0030
D	90.0	(52.8)		3.0	(8.8)		3.0	(8.1)	
B	28.2	13.3	0.8033	3.0	0	1.0	3.1	0.0	1.0
C	14.8	(52.8)		3.0	(8.8)		3.1	(8.1)	
B	28.2	61.8	0.2556	3.0	0	1.0	3.1	0.1	0.9903
D	90.0	(52.8)		3.0	(8.8)		3.0	(8.1)	
C	14.8	75.2	0.1702	3.0	0	1.0	3.1	0.1	0.9903
D	90.0	(52.8)		3.0	(8.8)		3.0	(8.1)	

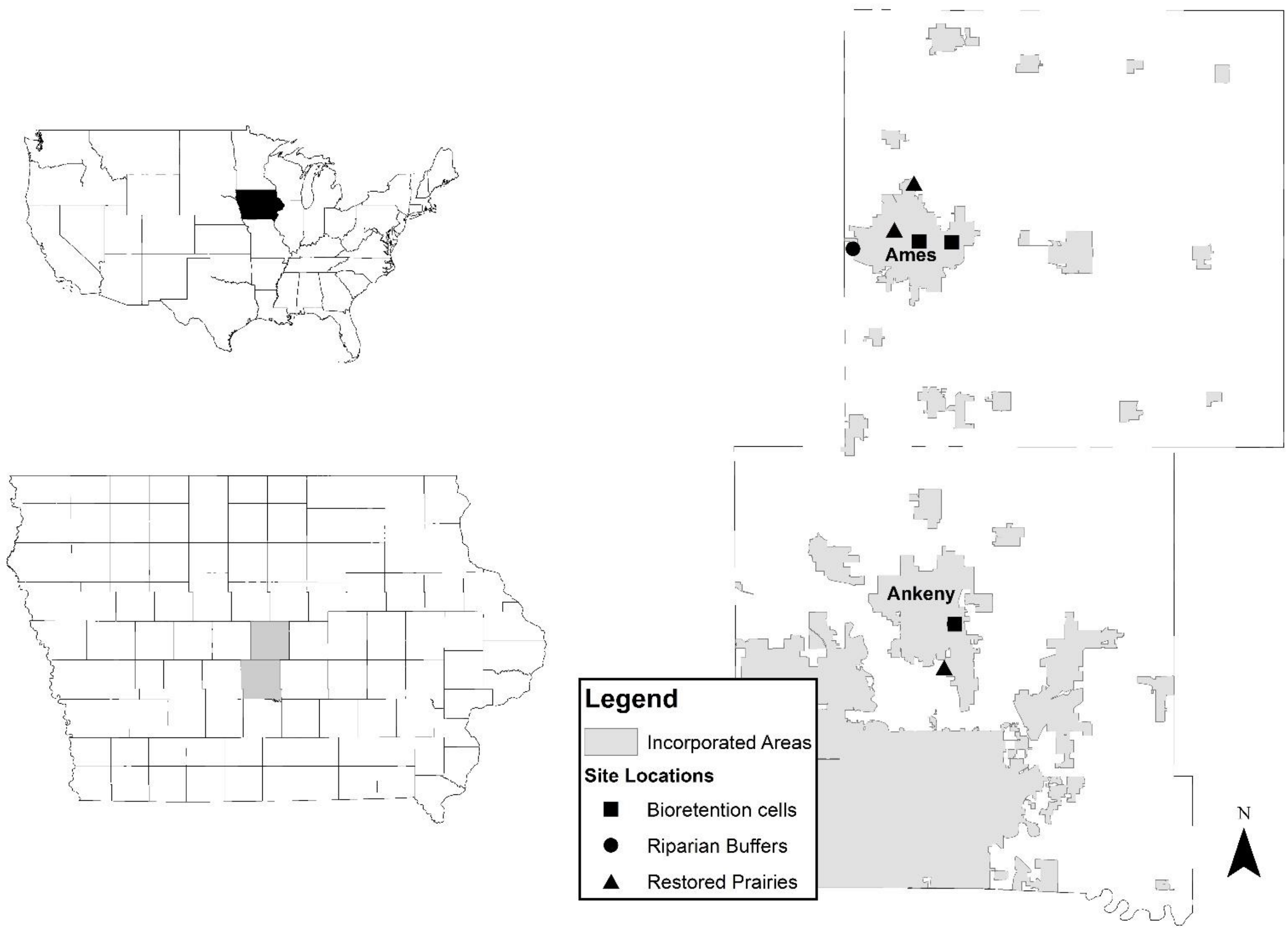

Figure 1: Generalized map of the United States; map of the state of Iowa and counties; incorporated boundaries of the City of Ames (Story County) and the City of Ankeny (Polk County) with locations of best management practices.

CHAPTER 3: MULTI-SPECIES VEGETATED RIPARIAN BUFFERS IN RURAL AND URBAN LANDSCAPES: DO THEY FUNCTION SIMILARLY?

A manuscript formatted to be submitted to Water, Air, & Soil Pollution

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Abstract: Streams are often degraded through human activities such as intensive agriculture and urban development, which deliver polluted runoff and sediments to aquatic systems. Streamside conservation practices, such as vegetated riparian buffers, have been designed to intercept and infiltrate runoff, pollutants, and sediments by improving soil structure and stability. The objective of this study was to assess performance of constructed three-zone multi-species riparian buffers in both rural and urban settings. Performance was measured by quantifying their spatial characteristics, soil properties, infiltration rates, time-to-runoff, and pollutant capture compared to adjacent contributing areas. Despite space constraints (narrower vegetated areas on both sides of the stream), urban buffers had larger surface area ratios compared to their contributing areas. In both rural and urban settings the buffer areas had the highest infiltration rates, and the wooded buffer zones demonstrated significantly greater time-to-runoff. An overall analysis of differences in infiltration parameters between the buffers and their contributing areas indicated that they functioned similarly in these two landscape contexts. We did not find any differences related to pollutant capture. Both rural and urban buffers are likely to provide adequate source control for runoff generated by frequent low-intensity rain events. Rather than use of generic guidelines, we recommend more site-specific buffer designs to meet specific remediation targets: where

infiltration is a primary objective, increasing the width of buffer wooded vegetation zones is likely to improve buffer performance, whereas for sediment and (sediment-attached) phosphorus capture increasing the size of buffer prairie zones is likely to contribute to better performance.

Keywords: Streamside conservation practices • Urban stormwater management • Infiltration rates • Watershed management • Best management practices

Introduction

Riparian ecosystems occur at the land-water interface along stream corridors and provide a number of ecosystem services: naturally intercepting and filtering water, sediments, and nutrients from adjacent landscapes before they reach the stream, providing wildlife habitat and food web maintenance, and improving recreational and aesthetic values of the landscape (Findlay and Taylor, 2006; Groffman et al., 2003; NRC, 2002). However, riparian systems are often impaired because of human activities causing changes to the land surface itself and the vegetation, as well as related alterations in hydrological flows and pollutant loads (Segura and Booth, 2010; Smucker and Detenbeck, 2014). In addition to clearing streambank vegetation, often the channels themselves are altered (e.g., straightened) to facilitate alternate land uses such as agricultural production or urban development (Kenwick et al., 2009).

A commonly used practice in intensive agricultural production, tillage creates vertical fissures in the soil profile to provide better drainage (Olson et al., 2013). However, the use of heavy machinery to conduct tillage often also negatively affects soil physical properties such as bulk density, pore size distribution, aggregate size distribution, and saturated hydraulic conductivity (Benjamin, 1993). Compaction, surface roughness, and other physical properties can also expedite erosion, sedimentation, and gully formation caused by uncontrolled runoff (Lindstrom, 1998). This typically leads to degradation of streams in the affected landscape.

Urban development also negatively affects natural hydrological functions. Expanding areas of impervious surfaces and construction activities that cause soil compaction increase the likelihood for delivery of polluted stormwater runoff to nearby streams. Natural vegetation in riparian corridors is often removed and replaced using ornamental plants that are not as effective for stabilizing stream channels (Booth and Jackson, 1997). Loss of forest canopy results in uncontrolled stream temperatures, lack of organic matter for aquatic habitats, and minimal in-stream debris to dissipate flow energy (Booth and Jackson, 1997; Groffman et al., 2003). At worst, urban stream channels may be buried or replaced entirely by concrete (Groffman et al., 2003; Sarvillinna et al., 2016), and suffer from what is known as the “urban stream syndrome” (Walsh et al., 2005).

One approach to protecting streams in both agricultural and urban landscapes is to restore vegetated riparian buffers along streambanks, which can slow runoff, improve soil structure, promote infiltration, and capture sediment and other non-point source pollutants (Lee et al., 2003). Constructed/vegetated riparian buffers are a best management practice recognized and supported by federal agencies through cost-share programs for private landowners (e.g., the Environmental Quality Incentive Program; USDA-NRCS, 2016). Incentive program guidelines and other technical guides describe recommendations for buffer design widths, planting schedules. Improving stream water quality has been shown to be a function of buffer width (Clinton, 2009; Mayer et al., 2005), which is arguably the most important controllable factor in buffer design (Gilliam, 1994). Greater buffer widths allow more space for vegetation to trap sediment (Richardson et al., 2012), and increase lag time and flow length. This increases the time available for infiltration to occur (Meals et al., 2010), and provides for temporal and spatial diffusion of pollutants (Meynendonckx et al., 2006). Although regional maintenance recommendations exist, most standards do not include a requirement of monitoring for water quantity or quality mitigation after buffer establishment (NRCS, 2012).

Interest in use of constructed vegetated riparian buffers in rural landscapes increased rapidly in the 1980s and 1990s because of their potential to intercept nitrogen and phosphorus in runoff from intensively managed agricultural fields (Cooper et al., 1987; Jacobs and Gilliam, 1985; Lowrance and Sheridan, 2005; Peterjohn and Correll, 1984). These pollutants were ultimately leading to the development of growing hypoxic zones in receiving waters such as the Gulf of Mexico (Carpenter et al., 1998). Streamside management systems such as reconstructed/vegetated buffers were initially proposed as practices that could be used alongside in-field conservation strategies to address these problems (Lee et al., 2003). Many studies have documented the effects of restored vegetated buffers in rural watersheds (Karr and Schlosser, 1978; Lowrance et al., 1984; Mahoney and Erman, 1984; Osborne and Kovacic, 1993). For example, researchers have demonstrated that buffer zones have the ability to remove up to 97% of sediment and 94% of nitrogen in agricultural runoff (Lee et al., 2003), retain up to 85% of phosphorus (Polyakov et al., 2005). Strong correlations between riparian root biomass and soil organic matter content and denitrification have also been documented (Gift et al., 2010).

Interest in riparian buffer restoration along urban streams has also increased, although this activity has been more recent than that in rural landscapes (Bernhardt et al., 2005). Despite interest in implementing urban stream buffers (e.g., Herringshaw et al., 2010), land ownership patterns and existing infrastructure (e.g. roads, buildings, and sewers) may limit opportunities to restore urban riparian corridors (Smucker and Detenbeck, 2014).

Although one might expect similar performance of analogous practices installed in rural and urban watersheds, to our knowledge few studies have been done that directly compare vegetated riparian buffers in rural and urban landscapes. In this study we examined the characteristics of constructed multi-species riparian buffers in both settings. Our research objectives were to answer the following questions: (1) Given the space limitations that are likely in urban landscapes, do urban riparian buffers have smaller practice area to contributing area ratios compared to rural buffers? (2) Are differences in infiltration rates and absorption capacity between buffer receiving areas and contributing landscape areas greater for urban buffers compared to rural buffers? and (3) Are there differences in the kinds and quantities of pollutants in runoff water and in soil for urban and rural buffers compared to their respective contributing areas?

Methods

Study area

The study sites included in this research were all located in the Des Moines Lobe ecoregion (Chapman et al., 2002) of central Iowa, a depositional remnant of the late Wisconsin glaciation located within the Upper Mississippi River basin (Simpkins et al., 2002). The landscape varies from flat to undulating slopes and contains soils that formed in loamy glacial till, local alluvium, or colluvium generated from till, and that range from well drained to poorly drained (Schultz et al., 1995). The landscape is dominated by intensive agriculture (approximately 87% is dedicated to rowcrop production; Simpkins et al., 2002) that began about 150 years ago. The urban areas in this landscape were also founded about 150 years ago. The two rural riparian sites we studied are located in northeast Story County, in central Iowa. The two urban riparian buffer sites we studied are located within the City of Ames and City of Ankeny, also both in central Iowa. Both cities are characterized by growing populations, which are estimated to have increased by 10% (Ames) and 25% (Ankeny) between 2010 and 2015. Population growth has led to rapid expansion of these urban areas and increases in their impervious surface coverage: recent studies indicate that the City of Ames has approximately 25% (city-wide average) impervious land cover (Jake Moore, City of Ames, personal commun.) and the City of Ankeny has approximately 19% (in 2011) average impervious land cover (Wu and Thompson, 2013).

Study sites

Rural riparian buffers

We assessed rural buffers located along Bear Creek and Long Dick (LD) Creek in central Iowa (Figure 1). The Bear Creek buffer is located on a third order stream in a research and demonstration project area established in 1990 (Schultz et al., 1995). This buffer extends along a stream reach of 1,000 meters within a privately owned rowcrop agriculture enterprise. Our observations were conducted in one 90-m subplot within the larger buffer. The second rural buffer is located along LD Creek in an area formally used as pasture land until the buffer was established in 2003 and placed in the Conservation

Reserve Program (Dick Schultz, Iowa State University, personal commun.; ISU GIS Support and Research Facility, 2017). Our observations were conducted in a 100-m meter reach of the LD Creek buffer. Both rural buffer sites were planted with zones of trees including silver maple (*Acer saccharinum L*.), green ash (*Fraxinus pennsylvanica Marsh*), and black walnut (Juglans nigra), shrubs including Nanking cherry (*Prunus tomentosa*) and red osier dogwood (*Cornus stolonifera Michx.)*, and prairie grass, primarily switchgrass (*Panicum virgatum*) (Schultz et al., 1995).

Urban riparian buffers

The urban riparian buffers were planted in 2007 (in Daley Park in Ames IA; described in Herringshaw et al. 2010) and in 2011 (in Summerbrook Park in Ankeny, IA; hereafter SB Park; Amy Bryant, City of Ankeny, personal commun.; also Figure 1). Each buffer contains a zone of native tree species including swamp white oak (*Quercus bicolor*), bur oak (*Quercus macrocarpa*), river birch (*Betula nigra*), American sycamore (*Platanus occidentalis*), and northern hackberry (*Celtis occidentalis*) planted closest to the stream, with an intermediate zone of native trees and shrubs such as ninebark (*Physocarpus opulifius*), and elderberry (*Sambucus canadensis*), and furthest from the stream a zone of native prairie grasses and forbs (such as big bluestem (*Andropogon gerardii*), Canada wild rye (*Elymus canadensis*), and side-oats grama grass (*Bouteloua curtipendula*) and forbs such as black-eyed Susan (*Rudbeckia hirta)*, yellow coneflower (*Ratibida pinnata*), false sunflower (*Helopsis helianthoides*) and vervain (*Verbena stricta*). In both cases the urban buffers run parallel to the stream and perpendicular to nearby roadways, and vary in width throughout the reach of stream due to spatial constraints in each park (e.g. sidewalks and other park amenities). Both are located in municipally-managed parks adjacent to residential neighborhoods. For these sites we made measurements along the entire length of each buffer.

Riparian buffer and contributing area delineations

All four riparian buffers and their contributing areas were delineated using a Geographic Information Systems (GIS) model. Light Detection and Ranging (LiDAR) data were used to generate digital elevation models (DEMs) at a one-meter resolution (Iowa LiDAR Mapping Project, GeoTREE, 2017). These models were used to determine subwatershed boundaries and dimensions for the area draining to each buffer from both sides of the stream. Land cover data were extracted from the Natural Resources GIS Library in order to determine the area of impervious cover in each of the subwatersheds (NRGIS, 2017).

Soil bulk density and volumetric water content

Surface soil samples were collected adjacent to the location of three infiltrometer tests (described below) at each buffer to determine soil bulk density and soil water content. Soil samples were collected using an AMS Bulk Density Compact Slide Hammer™ to extract a 90.4-cm^3 cylindrical sample from the first 7 cm of soil. Samples were sealed in plastic bags, chilled immediately and transported in a cooler to the laboratory for further processing.

Soil bulk density samples were weighed, oven-dried at 70˚ C for 48 hours, and weighed again to determine mass of water and mass of soil in sample. Bulk density was calculated as soil dry weight divided by sample volume. Volumetric water content, expressed in cm^3 cm^{-3} was calculated using the following equation where

$$Volumetric\ water\ content\ \left(\frac{cm^3}{cm^3}\right) = \frac{m_{water}/\rho_{water}}{m_{soil}/\rho_{soil}} = \frac{\theta_g * \rho_{soil}}{\rho_{water}}$$

Where $\frac{m_{water}}{\rho_{water}}$ = the mass of the water in the soil over density of water, which is divided by $\frac{m_{soil}}{\rho_{soil}}$ = mass dry soil over the density of soil, or the gravimetric water content θ_g multiplied by bulk density over the density of water. Subsamples (40 g) were then mixed thoroughly with a sodium-hexametaphosphate solution (50 g L^{-1}) and analyzed to determine particle size distribution using the hydrometer method (ASTM, 1985; Gee and Bauder, 1986).

Infiltration measurements

At each riparian buffer site, a set of three infiltrometer tests were conducted in each of the wooded buffer zones, the prairie buffer zones, and in the contributing areas between June and September, 2016. We used a portable single-ring Cornell Sprinkle Infiltrometer™ (van Es and Schindelbeck, n.d.) to determine field saturated infiltration rates, time-to-runoff, and sorptivity. The cylindrical water reservoir of the infiltrometer has a perforated bottom which delivers rainfall onto a 24.1cm (9.5 inch) diameter area controlled by a 20.3-cm (8-inch) tall metal ring (van Es and Schindelbeck, n.d.). Rainfall rate was calibrated by adjusting a Mariotte tube, which also controlled for constant head. Rainfall rate was calibrated each day to deliver 0.6 cm min^{-1} (0.24 in min^{-1}) using a two-minute test. Simulated rainfall rates were also determined directly to account for variation caused by field conditions (e.g., temperature variations). The reservoir was filled with deionized water transported from the laboratory, and the simulated rainfall rate was calculated by determining the difference in height of the water in the cylinder before and after the timed observation divided by the time elapsed.

The metal ring was fastened to the base of the reservoir and carefully inserted to be level with the soil surface to a depth of 7 cm. An outflow hole at 7 cm was set to be flush with the soil surface and was fitted with an effluent tube to allow water to run off once surface ponding occurred. We recorded rainfall rate, time-to-runoff, and volume of runoff water at three minute intervals. The runoff rate (cm min^{-1}) was calculated using the following equation (van Es and Schindlebeck, n.d.):

$$runoff\ rate = \frac{V_t}{(457.30 * t)}$$

where 457.30 (cm²) = area of the metal ring, t = time interval (min), and Vt = volume of water (ml) during time interval t. After reaching a steady state for runoff rate, the field-saturated infiltration rate was estimated by determining the difference between the applied rainfall rate and the rate of runoff, allowing direct comparisons among sites with different antecedent soil moisture contents (van Es and Schindlebeck, n.d).

Determination of buffer capacity and potential water storage

Urban stormwater design recommendations stipulate that practices be sized to capture/treat runoff from 90% of storms in a typical year, which are based on a rainfall depth of 3.18 cm hr^{-1} (1.25-in hr^{-1}) in central Iowa (Iowa DNR, 2009). However, given near-future climate scenarios that include more frequent intense rainfall events, we chose to estimate runoff generation for each site using a 5.08 cm hr^{-1} (2 in hr^{-1}) rainfall intensity. To estimate runoff for this hypothetical event, we subtracted the average infiltration rate (cm hr^{-1}) from this precipitation intensity and multiplied by the total surface area of each zone to determine the volume of water that would be generated (runoff, indicated by a positive value) or absorbed (infiltrated, indicated by a negative value) by each buffer zone and contributing area (Herringshaw et al., 2010).

Estimates for potential water storage volumes in the buffers were found by clipping the digital elevation model (DEM) raster to the size of the surface area of each buffer zone. The focal statistics tool (ESRI Spatial Analyst, n.d.) was used to create a new DEM of the minimum elevations within a 20 to 60 meter radius (depending on buffer width) to find the elevation of the stream bed along each stream reach. The difference between height of the water in the stream (uniformly set at 0.5 meters above the stream bed) and surface elevation within the buffer was summed over the entire surface to estimate the volume of soil contained in each buffer zone.

Sample collection and analysis

Runoff water samples

All water samples were collected from the infiltrometer effluent tube at the time of initial runoff and were immediately chilled for transport and subsequent cold storage in the laboratory. Samples for measurement of nitrate concentration were collected in acid-washed (5% sulfuric acid; 2-hour rinse) and acidified (200 µg concentrated sulfuric acid) 125-ml bottles and analyzed using automated colorimetry (Method 353.2, USEPA 1993a). Acid-washed bottles also treated in a phosphorous-free soap bath (2-hour rinse) were acidified and used to collect 125-ml samples for analysis of total phosphorus concentrations using semi-automated colorimetry (Method 365.1, USEPA, 1993b) extracted with an aliquot of aqueous ammonium persulfate (AQ2 Method, USEPA, 2011). Acid-washed bottles were also used to collect 75-ml samples for determination of chloride concentrations using the low-level amperometric titration method (Method 4500-Cl E, APHA 2005). Samples were analyzed in the Riparian Management Systems Laboratory in the Department of Natural Resource Ecology and Management (nitrate) or in the Water Quality Research Laboratory in the Department of Agricultural and Biosystems Engineering (total phosphorus and chloride) at Iowa State University.

Surface soil samples

Soil samples were also collected from the soil surface (to a 7-cm depth) at each buffer zone for determination of nutrient concentrations (nitrate, ammonium, and total phosphorus). These samples were placed in plastic-lined soil sample bags, chilled, and delivered to the Iowa State University Soil and Plant Analysis Laboratory where they were analyzed using KCl extraction and cadmium reduction detection methods for nitrate-nitrogen and ammonium, and the Mehlich-3 extraction and ascorbic acid spectrophotometer detection method for total phosphorus (NCRR, 2015). Samples for determination of soil metal concentrations (chromium and zinc) were placed in 250-ml wide-mouth glass jars, chilled immediately, and delivered to the State Hygienic Laboratory within five hours of collection. Samples were analyzed there using inductively-coupled plasma mass spectrometry (Method 6020A and Method 6010C, respectively; USEPA, 2000b).

Data and statistical analysis

We calculated means for characteristics (soil physical and chemical properties, infiltration tests) for each buffer zone using three samples for each zone (wooded, prairie, and contributing area) at each rural and urban site (means represent six measurements). To account for possible correlation among multiple tests/samples from each site, we used a linear mixed model fit by restricted maximum likelihood, and t-tests using Satterthwaite approximations for degrees of freedom to estimate and compare means for infiltration characteristics (average infiltration rate, time-to-runoff, and runoff volume) for the buffer zones and their contributing areas. Differences between buffer zones and their contributing areas were compared within rural and urban sites to examine similarities and differences for the buffers in relation to their respective contributing areas. The differences between means for rural buffer zones and their contributing areas were compared to the differences between means of urban buffer zones and their contributing areas using the following equation (showing wooded zones):

$$Estimate\ of\ Differences = (Rural\ Buffers_{wooded} - Rural\ Buffers_{CA}) - (Urban\ Buffers_{wooded} - Urban\ Buffers_{CA})$$

Estimates of time-to-runoff were converted to a log scale and were right-censored at 40 minutes (total test time) if 100% infiltration occurred. We used Student's t-tests for pairwise comparisons of means for water and soil chemical parameters for buffer zones compared to their respective contributing areas. All statistical analyses were conducted in R (R Core Team, 2017); we set $p \leq 0.05$ to declare significance.

Results and Discussion

Riparian buffer and contributing area characteristics

The rural buffers are located in subwatershed contributing areas with surface areas of 131 ha and 100 ha, which are approximately seven to eight times larger than the areas of the buffers themselves (Table 1). Impervious cover is minimal (less than 1%) at the rural buffer sites, comprised of one residential property and some gravel road surfaces. Subwatershed contributing areas of the two urban buffers had surface areas of 3.87 ha and 1.22 ha, which are approximately three times larger than the buffers, and contained much higher proportions of impervious surfaces, from 19 to 21% (Table 1). Although the rural buffers themselves were larger, their relatively small practice area to contributing area ratios may limit buffer capacity to provide infiltration and pollutant removal functions in these agricultural landscapes. The urban buffer to contributing area ratios were larger than expected, and likely enhance their overall function for stormwater infiltration, and sediment and pollutant removal.

Riparian buffer width is critical for buffer functions such as interception of sediment and enhanced infiltration of runoff into the soil before reaching the stream (Sweeney and Newbold, 2014). The rural buffers had widths ranging from 30 to 150 meters extending from the streambank to the buffer edge. The urban buffers were much narrower, ranging from 5 to 25 meters. Guidelines for buffer width vary, but generally suggest a minimum and fixed width of at least 30 m (100-ft) in either rural or urban environments (e.g. Blinn and Kilgore, 2001; IEPA, 2010; Johnson and Buffler, 2008; MDEQ, 2014; Schueler, 1995; USEPA, 2010). Recommendations based on fixed widths can facilitate verification of conservation program requirements (Richardson et al., 2012) but may make it more difficult to customize practices to fit available spaces, especially in urban settings. In both urban landscapes we examined, existing sidewalks and roadways limited the extent of the buffers. Because of such spatial limitations, denser vegetation and shallower slopes have been recommended for urban riparian buffer zones especially in locations with relatively high proportions of impervious cover that can create flashier runoff patterns and higher peak flows (Polyakov et al., 2005). Alternatives to designs based on fixed buffer widths include use of models to determine more hydrologically active areas that should have wider buffers (Agnew et al., 2006) or buffers that vary in width along any given reach to accommodate other landscape features (Gorsevski et al., 2008).

Soil bulk density and volumetric water content

There were no significant differences in mean soil bulk density between buffer zones and contributing areas in either landscape setting, although the contributing areas in both cases had the highest bulk densities that we measured (Table 2). However, we observed dissimilar patterns in the two landscapes for volumetric water content. In the rural buffers water content in both the wooded and prairie zones was significantly lower than their rural contributing areas ($p = 0.0119$ and $p = 0.0355$, respectively; Table 2), while in the urban sites wooded buffer zones had significantly higher volumetric water content values than their urban contributing areas ($p = 0.0345$). In the urban buffers, these differences may be related more to the relatively low topographic position and proximity to the stream of the wooded buffer zones, where subsurface flow probably contributes to consistently higher soil moisture levels (Bosch et al., 1994; Clinton et al., 2009).

Infiltration characteristics

There were no differences in average infiltration rates between either rural or urban buffer zones compared to their respective contributing areas (Table 3) although buffer zones consistently had the highest values for this parameter. In earlier work at one of the same rural sites investigators concluded that that infiltration rates were in fact higher in the buffer than in adjacent cultivated land (Bharati et al., 2001). Our observations followed a similar pattern, however our sampling intensity at that site was lower and our experimental design included different sites. Thus, our inability to detect a significant difference could be due to our limited within-site sample size and a higher degree of variation between sites for this parameter.

For time-to-runoff, the wooded zone of both the rural and urban buffers was significantly higher than that of their contributing areas (Table 3). For runoff volume, negative estimates of differences indicate that contributing area values were high relative to those of the buffer zones, although these differences were not significant. We expected to see greater differences between the urban buffer zones and their contributing areas (compared to the degree of differences between the rural buffers and their contributing areas), based on the likelihood of greater soil compaction and its effect on infiltration characteristics in the urban contributing areas. However, the buffers in both types of landscape functioned very similarly, and only the wooded buffer zones demonstrated greater time-to-runoff. In the case of the urban buffers, their relatively small size and shorter time since establishment may be limiting their infiltration functions relative to their contributing areas (Smucker and Detenbeck, 2014).

We also analyzed the differences between means of rural buffer zones and their contributing areas compared to the differences between means of urban buffer zones and their contributing areas (e.g., the difference of the differences; Table 4). Negative estimates for the prairie zones indicate that mean differences between those zones and their contributing areas was greater for the urban buffers. Overall, lack of differences in this analysis corroborates the finding that the buffers in both landscape contexts are performing similarly in spite of variation in their sizes and locations. However, standard errors are also quite high, possibly due to characteristics of the terrain and/or soil properties which are also likely to cause variability in these infiltration parameters (e.g., Dosskey, 2002).

Buffer capacity and potential water storage

The rural buffers as well as their contributing areas were estimated to absorb all incident precipitation from a hypothetical 5.08 cm hr^{-1} (2 in hr^{-1}) rain event (Table 5). Either higher antecedent moisture content and/or a more intense rainfall event would be necessary to generate runoff to the buffers from their surrounding contributing areas. The urban riparian buffers also had adequate capacity for this event, as well as for runoff generated from their contributing areas (Table 5). Based on this rain event only the contributing area of the Summerbrook Park buffer generated runoff.

Estimates of potential volume of water held in soil found by multiplying volumetric water content by volume of soil did not reveal distinct patterns among the buffers (Table 6). These estimates were much greater for the rural riparian buffers, probably due to the much larger soil volumes within their practice areas. Variation in volumetric water content at the time of sampling had a strong effect on this analysis – for example, moisture content was much higher in the Summerbrook Park buffer at the time of sampling, which led to the relatively high volume we calculated for water storage at that site.

Sample characteristics

Runoff water samples

There were no differences in nitrate or chloride concentrations of the effluent runoff water for the buffer zones compared to their contributing areas for either the rural or the urban buffers (Table 7). Total phosphorus concentrations in runoff were higher in the rural prairie buffer zones than the rural contributing areas ($p = 0.0386$). Because the prairie buffer zones are adjacent to the contributing areas, their relatively dense above ground vegetation functions as a sediment trap, so it is likely that sediment-attached phosphorus is accumulating in this zone.

Surface soil samples

We did not detect significant differences for nutrient or metal concentrations in surface soil samples from the buffer zones and their respective contributing areas in either landscape context (Table 8). Average total phosphorus concentrations were highest in the rural contributing areas and probably reflect fertilization practices. Although contributing areas in both landscapes had the highest average ammonium concentrations there were no significant differences compared to their respective buffer zones.

Because different mechanisms dominate nitrogen and phosphorus movement and export, different patterns in water and soil concentrations occur for these two nutrients in both landscapes. Nitrate is mainly transported through the watershed via subsurface flow allowing for greater spatial and temporal diffusion (Pionke et al., 2000). In contrast, phosphorus is typically associated with sediment transported by overland flow, and thus is more likely to be measured in surface soil and water sampling (Lee et al., 2000). Evidence from previous studies indicates that soil nutrient and metal concentrations may vary widely throughout a buffer because of soil drainage characteristics within the riparian area rather than being related to surrounding landscape use (Mayer et al., 2005; Norton and Fischer, 2000; Parkyn et al., 2003).

Summary and Conclusions

Restored vegetated riparian buffers are well-recognized as a best management practice to address potential non-point source pollution in agricultural watersheds. However, their use in urban landscapes has been more recent and is limited. Existing gray infrastructure, such as conventional sewer systems and expansive impervious surface cover may reduce the area available for similar buffers in urban landscapes. The functional capacity of constructed vegetated riparian buffers to protect stream ecosystems and provide more natural in-stream conditions may be lower in urban riparian landscapes, although noticeable improvements in water interception and enhanced stream ecosystem condition are likely over time.

Our study indicates that restored vegetative riparian buffer areas may function similarly in rural and urban environments. The urban riparian buffers were much smaller (both length and width) but had greater practice to contributing area ratios, indicating the potential for capture of runoff and pollutants in these “downscaled” landscape segments. The rural buffers and their subwatershed areas were both much larger, but the ratios between buffer receiving areas and the contributing landscape areas were much smaller, indicating that the rural buffers should be designed to mitigate greater volumes of runoff from greater distances.

We determined that even though urban buffers may be limited in size, infiltration performance was similar to rural buffers. Our results indicate that both rural and urban buffers are likely to provide adequate runoff source control for frequent low-intensity rain events. We observed little variation in pollutant accumulation in either type of buffer we studied, and suggest that future research could investigate this aspect of their performance more intensively. We also acknowledge that in both landscape contexts the buffers may be bypassed by existing tile drains or storm sewer lines, which would limit the degree to which the restored buffer areas could address potential runoff and pollution problems.

Our results also point toward the need for customized buffer designs. Where infiltration is a primary objective, increasing the size of wooded vegetation zones is likely to improve buffer performance, whereas for sediment and (sediment-attached) phosphorus capture increasing the size of the prairie zones is likely to contribute to better performance. Overall, effective riparian buffer design requires whole-watershed-level consideration with site-specific remediation targets. Although urban settings often have space limitations, municipal planners and stormwater managers should consider implementing vegetated buffers to protect urban streams to the extent possible.

Acknowledgements

Support for this study was provided by the Graduate Program in Environmental Science at Iowa State University, the Iowa Department of Agriculture and Land Stewardship (State Soil Conservation Committee), the Center for Global and Regional Environmental Research, and the National Laboratory for Agriculture and the Environment. We thank the City of Ames, the City of Ankeny, and Ron Risdall for allowing us to use their land as research sites. We would like to thank Iowa State University's Statistics Consulting Lab, particularly Eric Mittman, for assistance with analysis. We thank Bret Lang, Breanna Marmur, Matthew Kessler, and Sarah Rueger for field and lab assistance.

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Table 1: General characteristics of sites where riparian buffers were evaluated in rural and urban landscapes in central Iowa, USA. Surface areas of management practices of four sites and their respective contributing areas by landscape type (rural or urban), ratios of practice areas to contributing area, and total impervious surface cover (roads, other pavement, roofs; by area and percent) within each contributing area.

Buffer zones	Location	Surrounding land use	Year	Surface area, ha	Practice area to CA ratio	Width range, m	Stream reach length, m	Impervious cover, m² (%)
Rural riparian buffers								
Bear Creek buffer	Rural Story County	Agriculture	1990	17.23	1 : 8	30-150 m	1000 m	
Bear Creek wooded buffer zone				9.04	1 : 14			
Bear Creek prairie buffer zone				8.19	1 : 15			
Contributing area				131.00				279 (1%)
LD Creek buffer	Rural Story County	Agriculture	2003	15.00	1 : 7	60-140 m	800 m	
LD Creek wooded buffer zone				8.65	1 : 12			
LD Creek prairie buffer zone				6.35	1 : 15			
Contributing area				10.00				0 (0%)
Urban riparian buffers								
Daley Park buffer	City of Ames	City park	2007	1.22	1 : 3	10-50 m	310 m	
Daley Park wooded buffer zone				0.81	1 : 5			
Daley Park prairie buffer zone				0.42	1 : 9			
Contributing area				3.87				8236 (21.3%)
SB Park buffer	City of Ankeny	City park	2011	0.38	1 : 3	5-25 m	170 m	
SB Park wooded buffer zone				0.24	1 : 6			
SB Park prairie buffer zone				0.14	1 : 10			
Contributing area				1.23				2349 (19.1%)

Table 2: Soil bulk density (g cm^{-3}) and volumetric water content (cm^3 cm^{-3}) for rural and urban riparian buffers and their contributing areas. Mean differences between buffer zones and contributing areas, their standard errors, and p-values for comparisons were calculated based on student's t-tests.

Buffer zones		Soil bulk density (g cm^{-3})			Volumetric water content (cm^3 cm^{-3})		
	n	Mean	Estimate of difference (Std. error)	p-value	Mean	Estimate of difference (Std. error)	p-value
Rural riparian buffers							
Wooded buffer zones	6	1.26	0.07 (0.12)	0.5547	0.19	0.09 (0.03)	0.0119
Prairie buffer zones	6	1.13	0.20 (0.12)	0.1128	0.21	0.07 (0.03)	0.0355
Contributing areas	6	1.33			0.28		
Urban riparian buffers							
Wooded buffer zones	6	1.18	0.22 (0.11)	0.0657	0.38	0.14 (0.06)	0.0345
Prairie buffer zones	6	1.32	0.08 (0.11)	0.5120	0.27	0.03 (0.06)	0.6580
Contributing areas	6	1.40			0.24		

Table 3: Estimates of differences between means of buffer zones (wooded and prairie), and respective contributing areas for average infiltration rate (cm hr^{-1}), time-to-runoff (minutes, right-censored), and runoff volume (milliliters) for rural and urban sites.

Buffer zone	n	Mean	Estimate of difference	Std. error	df	t ratio	p-value
Average infiltration rate, cm hr^{-1}							
Rural wooded buffer zones	6	36.9	15.5	8.8	2.0	1.7	0.2196
Rural prairie buffer zones	6	27.1	5.7	14.7	2.0	-0.4	0.7378
Rural contributing areas	6	21.4					
Urban wooded buffer zones	6	19.9	14.9	8.8	2.0	1.7	0.2307
Urban prairie buffer zones	6	14.4	9.4	14.7	2.0	-0.7	0.5873
Urban contributing areas	6	5.0					
Time-to-runoff, minutes (log-scale)							
Rural wooded buffer zones	6	3.4	2.2	0.4	28.0	4.9	<0.0001
Rural prairie buffer zones	6	1.9	0.7	1.1	2.0	-0.6	0.5917
Rural contributing areas	6	1.2					
Urban wooded buffer zones	6	1.1	1.5	0.4	28.0	3.4	0.0018
Urban prairie buffer zones	6	1.4	1.8	1.1	2.0	-1.6	0.2414
Urban contributing areas	6	-0.4					
Runoff volume, ml							
Rural wooded buffer zones	6	119.0	-3671.0	1375.0	3.0	-2.7	0.0748
Rural prairie buffer zones	6	2643.0	-1147.0	3530.0	2.0	0.3	0.7759
Rural contributing areas	6	3791.0					
Urban wooded buffer zones	6	4606.0	-3235.0	1375.0	3.0	-2.3	0.0991
Urban prairie buffer zones	6	6913.0	-929.0	3530.0	2.0	0.3	0.8170
Urban contributing areas	6	7842.0					

Table 4: Estimates of differences between means of rural buffer zones and their contributing areas compared to the differences between means of urban buffer zones and their contributing areas for average infiltration rate (cm hr^{-1}), time to runoff (minutes, right-censored), and runoff volume (milliliters) for rural and urban sites.

Differences of differences	Estimate of difference	Std. error	df	t ratio	p-value
Average infiltration rate, cm hr^{-1}					
Wooded buffer zones	0.5	12.5	2.0	0.0	0.9694
Prairie buffer zones	-3.8	20.8	2.0	0.2	0.8727
Time-to-runoff, minutes					
Wooded buffer zones	0.7	0.6	28	1.1	0.2795
Prairie buffer zones	-1.1	1.5	2.0	0.7	0.5501
Runoff volume, ml					
Wooded buffer zones	-436.0	1945.0	3.0	-0.2	0.8369
Prairie buffer zones	-218.0	4993.0	2.0	0.0	0.9691

Table 5: Estimate of absorption capacity riparian buffer zone and their contributing areas assessed at both the rural and urban sites in central Iowa, based on average infiltration rates, surface areas, for a hypothetical rain event of 5.08 cm hr^{-1} (2 in hr^{-1}). To estimate runoff for this hypothetical event, we subtracted the average infiltration rate (cm hr-1) from this precipitation intensity and multiplied by the total surface area of each zone to determine the volume of water that would be generated (total runoff, indicated by a positive value) or infiltrated (total absorption, indicated by a negative value) by each BMP and contributing area.

Buffers	Reach length, m	Average infiltration rate, cm hr^{-1}	Surface area, ha	Impervious surface area, m^2 (% of total)	Runoff generation, $m^3\ hr^{-1}$	Total absorption, $m^3\ hr^{-1}$
Rural riparian buffers						
Bear Creek wooded zone	1000 m	34.0	9.0			-26170.0
Bear Creek prairie zone		13.3	8.2			-6740.0
Contributing area		26.3	131.0	279 (1%)		-277830.0
LD Creek wooded zone	800 m	39.7	8.7			-29960.0
LD Creek prairie zone		40.8	6.4			-22670.0
Contributing area		16.5	100.0	0 (0%)		-114550.0
Urban riparian buffers						
Daley Park wooded zone	310 m	29.9	0.8			-2008.8
Daley Park prairie zone		6.2	0.4			-45.9
Contributing area		5.7	3.9	8236 (21.3%)		-236.2
Summerbrook Park wooded zone	170 m	9.9	0.2			-116.4
Summerbrook Park prairie zone		21.3	0.2			-241.8
Contributing area		4.3	1.2	2349 (19.1%)	90.0	

Table 6: Individual buffer zones (rural and urban) water holding capacities based on estimated soil volume at site. Volumetric water content ratio multiplied by the volume of soil in practice to find the total water holding capacity of buffer zone.

Urban and rural buffer zone	Reach length	Volumetric water content, $cm^3\ cm^{-3}$	Volume of soil in practice, m^3	Volume of water held in soil profile, m^3
Rural riparian buffers				
Bear Creek wooded zone	1000 m	0.19	108700.00	20653.00
Bear Creek prairie zone		0.26	136340.00	35448.40
LD Creek wooded zone	800 m	0.22	98200.00	21604.00
LD Creek prairie zone		0.24	70230.00	16855.20
Urban riparian buffers				
Daley Park wooded zone	310 m	0.29	1949.88	581.84
Daley Park prairie zone		0.25	3352.50	820.69
Summerbrook Park wooded zone	170 m	0.53	2862.63	1506.88
Summerbrook Park prairie zone		0.33	3202.02	1065.95

Table 7: Means for rural and urban stormwater management buffer zones (prairie and wooded) and contributing areas for effluent runoff water nitrate, total phosphorus, and chloride concentration comparisons to respective landscape contributing areas for each zone are based on p-values calculated for pairwise comparison using student's t-tests.

Effluent runoff concentration	n	Mean	Difference of means	Std. error	p value
Nitrate, mg kg^{-1}					
Rural wooded buffer zones	1	0.00	0.02	0.16	0.9151
Rural prairie buffer zones	3	0.08	0.06	0.11	0.5594
Rural contributing areas	6	0.01			
Urban wooded buffer zones	6	0.28	0.11	0.09	0.2374
Urban prairie buffer zones	5	0.15	0.02	0.09	0.8035
Urban contributing areas	6	0.17			
Total phosphorus, mg kg^{-1}					
Rural wooded buffer zones	1	0.15	0.11	0.32	0.7344
Rural prairie buffer zones	3	0.72	0.46	0.21	0.0386
Rural contributing areas	6	0.26			
Urban wooded buffer zones	6	0.30	0.26	0.17	0.1383
Urban prairie buffer zones	5	0.34	0.22	0.18	0.2321
Urban contributing areas	6	0.56			
Chloride, mg kg^{-1}					
Rural wooded buffer zones	1	0.56	0.43	1.60	0.7918
Rural prairie buffer zones	3	1.45	0.47	1.05	0.6626
Rural contributing areas	6	0.99			
Urban wooded buffer zones	6	0.95	1.43	0.86	0.1099
Urban prairie buffer zones	5	1.66	0.72	0.89	0.4294
Urban contributing areas	6	2.38			

Table 8: Means and differences of means for surface soil sample concentrations of nitrate, ammonium, total phosphorus, zinc, chromium, and total extractable hydrocarbons. P-values for comparisons between buffer zones and their contributing areas based on pairwise student's t-tests for both rural and urban riparian buffers.

Soil nutrient or metal concentrations	n	Mean	Difference of means	Std. error	p value
Nitrate, mg kg^{-1}					
Rural wooded buffer zones	6	1.3	2.7	1.3	0.1348
Rural prairie buffer zones	6	1.9	2.0	1.3	0.2222
Rural contributing areas	6	3.9			
Urban wooded buffer zones	6	2.1	0.8	8.5	0.7617
Urban prairie buffer zones	6	7.2	4.4	12.0	0.1687
Urban contributing areas	6	2.9			
Ammonium, mg kg^{-1}					
Rural wooded buffer zones	6	3.2	1.7	0.7	0.0972
Rural prairie buffer zones	6	2.9	2.1	0.7	0.0638
Rural contributing areas	6	4.9			
Urban wooded buffer zones	6	4.4	1.3	5.4	0.3910
Urban prairie buffer zones	6	4.5	1.2	5.3	0.4415
Urban contributing areas	6	5.7			
Total phosphorus, mg kg^{-1}					
Rural wooded buffer zones	6	92.5	110.5	74.9	0.2367
Rural prairie buffer zones	6	102.5	100.5	74.9	0.2723
Rural contributing areas	6	203.0			
Urban wooded buffer zones	6	49.0	1.0	19.3	0.9619
Urban prairie buffer zones	6	100.5	50.0	19.3	0.0810
Urban contributing area	6	50.5			
Zinc, mg kg^{-1}					
Rural wooded buffer zones	6	37.5	2.0	11.4	0.8718
Rural prairie buffer zones	6	34.5	5.0	11.4	0.6905
Rural contributing areas	6	39.5			
Urban wooded buffer zones	6	110.5	68.5	57.3	0.3175
Urban prairie buffer zones	6	47.5	5.5	57.3	0.9295
Urban contributing areas	6	42.0			
Chromium, mg kg^{-1}					
Rural wooded buffer zones	6	13.0	3.0	2.5	0.3081
Rural prairie buffer zones	6	12.0	4.0	2.5	0.2010
Rural contributing areas	6	16.0			
Urban wooded buffer zones	6	9.3	6.7	4.9	0.2635
Urban prairie buffer zones	6	14.5	1.5	4.9	0.7787
Urban contributing areas	6	16.0			

Figure 1: Generalized map of the United States; map of the state of Iowa and counties; incorporated boundaries of the City of Ames (Story County) and the City of Ankeny (Polk County) with locations of urban and rural riparian buffers.

CHAPTER 4: GENERAL CONCLUSIONS

Best management practices (BMPs) that include soil and plant components designed to collect, infiltrate, store and process precipitation could provide innovative, cost effective, and environmentally preferable solutions for hydrological management. However, adoption of these practices is not widespread, owing to a number of possible concerns related to their degree of effectiveness, perceived costs, and/or lack of incentives for their use. Vegetated infiltration BMPs were the focus of my studies because they could be easily integrated into both municipal and privately-owned areas and represent examples of small-scale practices that can be used alone or in combination with other practices to intercept runoff close to the source. Although some sites may not be appropriate for application of infiltration practices (e.g., highly contaminated areas where infiltrated water would pose a risk for groundwater contamination), control strategies could be designed for inclusion in new development or as retrofit structures in a variety of other landscape contexts to reduce the hydrological impacts of runoff on watershed areas, streams, and downstream surface waters. For this thesis I conducted two studies to examine and describe a set of practices that have been implemented on a limited basis in both urban and rural landscapes to evaluate their effectiveness for infiltration and pollutant capture.

Capture of stormwater runoff and pollutants by three types of urban best management practices

For the first study, a set of three bioretention cells, three areas of native landscaping (restored prairie), and two three-zone urban riparian buffers were examined. All of these practices had been successfully retrofitted into existing urban infrastructure.

On average, the bioretention cells and the buffer wooded zones had significantly lower soil bulk density, higher infiltration rates, and smaller runoff volumes than their contributing areas, and time-to-runoff was also significantly higher for the bioretention cells. However, based on practice capacity, the bioretention cells in particular may have been undersized relative to their contributing areas and in view of likely changes in anticipated precipitation patterns (more frequent intense rain events). Infiltration characteristics of the restored prairie areas were not significantly different from those of the contributing areas, suggesting that soil treatment (i.e., compost addition) before establishment of the vegetation could enhance their water infiltration and storage functions. In a detailed assessment of one bioretention cell, I found evidence for effective capture and storage of nutrient, metal, and hydrocarbon pollutants. My findings indicate that more widespread implementation of these practices would reduce stormwater runoff and lead to improvements in surface water quality.

Multi-species vegetated riparian buffers in rural and urban landscapes: do they function similarly?

In this study, I measured the performance of constructed multispecies vegetated riparian buffers, two each in rural and urban settings, in relation to their contributing areas. Despite their smaller size, urban buffers had larger surface area ratios within their subwatersheds than did rural buffers. Although average infiltration rates were highest in the buffer areas, I did not detect significant differences in infiltration compared to their respective contributing areas. The wooded buffer zones, however, demonstrated significantly greater time-to-runoff. Overall analysis of the hydrological performance of the two sets of buffers compared to their contributing areas suggested that these practices do perform analogous functions in these two landscape settings. We found few differences related to pollutant capture. In lieu of generic guidelines for buffer structure, I suggest that site-specific designs are more likely to meet specific goals dictated by landscape context.

Well-planned landscape scale integration of BMPs for runoff interception could improve the hydrological characteristics of both rural and urban landscapes, and vegetated practices in particular could support a range of additional benefits such as energy savings, mitigation of the urban heart island effect, aesthetically-pleasing landscape features, and wildlife habitat. Continued implementation of "demonstration" facilities especially by municipal staff in urban areas could increase awareness, understanding, and social acceptance of runoff source control practices which may lead to more widespread use on privately owned and managed lands. Continued monitoring of BMPs and greater accessibility to monitoring data would benefit future research and support increased implementation of these practices. Overall, the promotion and integration of vegetated BMPs in the early stages of site planning and design could lead to more sustainable and multifunctional landscapes.